APPROPRIATE MODELING FOR INTEGRATED FLOOD RISK ASSESSMENT

Yan Huang

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by

Yan Huang

born on September 25, 1971

in Guizhou, China

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In memory of my mother, Luo LaiQing

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Yan Huang

28 October 2005
Enschede, The Netherlands

Contents

Chapter 1

Introduction

In this chapter a brief introduction is given on issues that play a role in Integrated River Basin Management (IRBM). Aspects related to the design and application of Decision Support System (DSS) for IRBM are also discussed. A state of the art analysis seems to indicate that there is a gap between scientific knowledge available and its implementation in DSS system design when it comes to present day River Basin Management (RBM) applications. Similar trends are also observed in the DSS development for Integrated Flood Risk Assessment (IFRA), which is one of the core components of an integrated DSS. The general background on these issues is provided, specific thesis objectives and research questions are formulated, and the outline of this dissertation is presented.

1.1 Background

There is a water crisis that involves sustainability and integrity of water resource management (Cosgrove and Rijsberman, 2000; HRH, 2002; Van der Veeren and Lorenz, 2002). Human society has threatened its water resources resulting in a world wide water crisis (UN/WWAP, 2003). Many countries are already suffering a water crisis that affects many people and their ecosystems. Over 1 billion people are lacking access to safe drinking water, more than 3 billion lack access to sanitation. With current practices, the degradation of ecosystems and the loss of biodiversity will threaten the lives of future generations. It is clear that a sustainable management of water resources is needed.

In addition to sustainability of water resources, recently flood risk has gained increasing attention recently. Flood disasters are among the world's most frequent and damaging types of disasters and a trend is observed of increasing frequency of occurrence. During the latter half of the 20th century floods were the most common type of geophysical disaster, generating over 30% of all disasters between 1945 and 1986 (Glickman et al., 1992). These estimates are corroborated by more recent data from Munich Reinsurance for the period 1986-1995 (United Nations, Department of Humanitarian Affairs, 1997). One way of scaling the world's flood problem is to examine estimates of the number of people and properties located in (or exposed to) flood-prone areas. Some estimates (Parker, 1996) have been produced for a small number of countries revealing widely varying proportions of total country populations which are flood-prone. Typical numbers are 3.5% in France, 4.8% in the United Kingdom, 9.8% in the United States, over 50% in The Netherlands and 80% in Bangladesh. To minimize the potential damage, or to mitigate the flood risk either by short-term flood management (for example evacuation), or by long-term planning (for example by constructing flood defense systems) Flood Risk Assessment (FRA) is essential.

However, it has been found that the current FRA approaches, including risk-analyses either based on a statistical approach or on a physically-based approach, are incomplete since they neglect of the effect of flow velocity. This is not a real problem for relatively flat areas when the flow velocities in case of inundation are low. However, in hill slope areas, or when the flow velocities are high (e.g. due to the occurrence of a river dam break or an ocean tsunami), considerable damage can be incurred by large flow velocities, jeopardizing property and even human lives.

In a conventional and FRA approach the damage functions are often based on simple water depth ~ damage curves, and inundation modeling can be reasonably simplified by using a mass-balance approach, which is greatly facilitated nowadays by making use of the rapidly increasing capabilities of Geographic Information System (GIS) technology. However, such an approach cannot easily account for the momentum characteristics of actual flows, and will hence miss the important effect of flow velocity. In order to properly account for the physical principles, detailed hydrodynamic (hydraulic) modeling is required, taking into consideration the hydraulic characteristics of a river and its floodplains. Examples of this type of modeling, notably the combined one- and two-dimensional capabilities of the SOBEK1D2D system of WL | Delft Hydraulics, are presented in the chapters 4 and 5 of this thesis. Since the driving forces in flood simulation are largely determined by the elevation data and roughness information (which nowadays is commonly contained in GIS format), one of the questions addressed in this thesis is whether an a-priori estimation / approximation of the expected extreme flow velocities can already be obtained from the digital elevation and land use data, without actually carrying out full hydrodynamic simulations. This approach may be very suitable for rapid FRA, when timing is crucial and data availability is limited. So far such approximation has not been observed in the scientific literature.

FRA is not the only issue in IRBM. Due to the presence of multiple objectives, sometimes even conflicting interest of various stakeholders, and the *interaction* between the different processes, IRBM has a broad scope: on the one hand protecting mankind against the dangers of flooding, on the other hand preserving water for dry seasons and at the same time assuring other water functions like shipping and irrigation. This calls for long-term, IRBM strategies, the complexity of which makes computer-based tools like integrated Decision Support Systems, a necessity.

1.1.1 Integrated River Basin Management (IRBM)

The beginning of IRBM can be traced back to the creation of the Tennessee Valley Authority in the 1930s (Creighton, 1999). In 1956 the Economic and Social Council required its staff and a panel of experts to prepare a review of the administrative, economic and social implications of integrated river basin development. The document was widely discussed and re-issued in revised form in 1970 (UN Economic and Social Affairs, 1970). Their appraisal covered a broad range of problems associated with RBM, including land use surveys, economic evaluation methods, health implications and the Helsinki Rules on the Use of Water of International Rivers. The concept of *integration* has received increasing attention ever since (Downs et al., 1991; Welp, 2001; Van Ast, 1999; Angelakisa and Bontouxc, 2001; Van der Veeren and Lorenz, 2002; Tsagarakis et al., 2003). Tippett (2005) stated that for sustainable RBM, a *whole view* (others refer to this as a *holistic view*) shall be taken through the study of dynamic processes and the emergence of properties of *wholes* at different scales. This implies a need for integration due to the nature of rivers, comprising multiple physical, socioeconomic, and ecological processes, as well as the broad scope of RBM involving various inter-related activities, such as planning, construction and operation. Moreover, there always remains the gap between *advancing scientific knowledge* and *addressing practical needs*. Since its emergence in the early 30's of the 20th century, Integrated Water Resources Management / Integrated River Basin Management has been recognized as a worldwide issue. Numerous efforts have been made to obtain effective approaches. It seems widely agreed that the challenge lies in bridging the gap between science and practice (Westcoat, 1992; Nienhuis et al., 1998; Jonker, 2002; HRH, 2002; Jeffrey and Gearey, 2003).

Integrated river basin management can be defined as *the process of combining available data and disciplines using supporting tools and engineering algorithms to effectively formulate a strategy.* This definition emphasizes that IRBM is not only a goal in itself, but a process of balancing conflicting interests. As a process, integration in IRBM aims at a systematic approach to analyze and define problems and their solutions properly, resolving conflicting interests among the different components involved, as shown in Figure 1.1. Because of this complexity, IRBM is a complicated task.

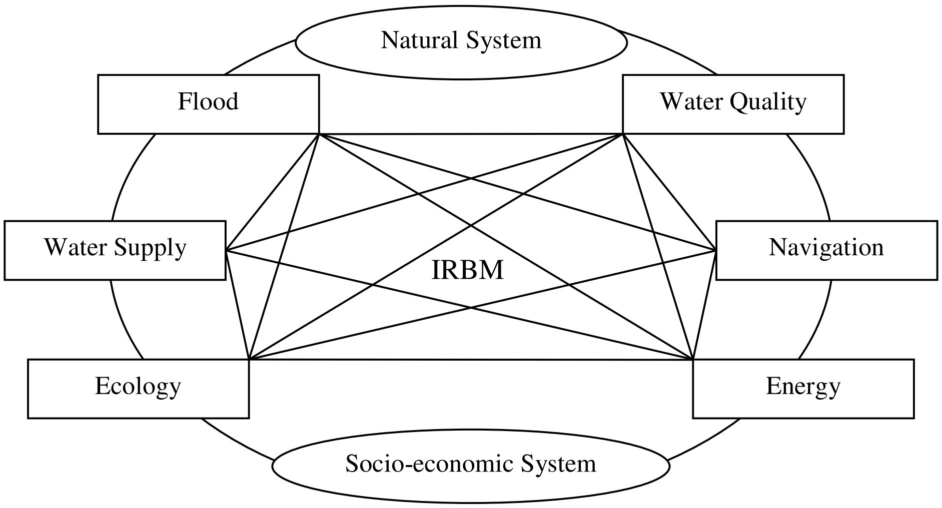

Figure 1.1 *Cross-links between various interests related to the natural system and the socio-economic functions of a river basin*

A river basin can be defined as the geographical area demarcated by the hydrological limit of the system of water, including surface and subsurface water, flowing into a common point (the groundwater may, however, not exactly follow this rule; Jain and Singh, 2003). Hence IRBM quite naturally involves multiple disciplines such as hydrology, hydraulics, ecology, water quality, navigation, but also disciplines like e.g. socio-economics and risk assessment. As a consequence, communication between the various stakeholders requires considerable attention. Decision Support Systems can help in this communication by facilitating different views from different angles, and by visualizing the effects and implications of potential measures. For example, the implementation of a flood mitigation measure such as the use of an upstream retention basin, should only occur when there is an effective reduction of the downstream risk. In such analysis, both natural system constraints (such as the storage volume of the retention basin) and the side effects (the minimum damage a particular measure would cause) are to be accounted for. However, flooding a certain area might harm its ecological system; in such a case, the objective of minimizing the negative impacts on the ecosystem – represented by certain ecological indicators – should be taken into account as well. Similar problems may occur for other components such as water quality and water supply.

Moreover, the broad scope of activities introduces a high level of complexity and difficulty for IRBM. This can be explained by considering the six different activities distinguished in RBM by Jain and Singh (2003). These are: (i) planning, (ii) construction, (iii) operation, (iv) monitoring, (v) analysis, and (vi) decision making. Each activity has a cause-effect relationship with others. Planning and construction are primary means to install facilities for operation and management which affect rivers in many ways, while monitoring and analysis provide inputs for planning, construction and operation. For example, to prepare the construction of e.g. a temporary water storage basin, different inputs are needed on physical, ecological, socio-economic and financial issues. Apart from the environmental impact assessment that is nowadays often legally required in the decision making process, public participation also has to be taken into consideration. For long-term sustainability, it is necessary to improve the awareness and knowledge of river users so that their participation can become more relevant and useful; continuing education for end users and the general public should play a vital role in IRBM.

Integrated management requires comprehensive insight in potentially conflicting interests (Nunneri and Hofmann, 2004). Consequently, IRBM has to deal with complexity and uncertainty in an effective way. Although considerable effort has been spent on improving the effectiveness of IRBM, this remains a problematic area due to the gap between theory and practice (Westcoat, 1992; Nienhuis et al., 1998; Jonker, 2002; Jeffrey and Gearey, 2003). The difficulties involved in IRBM are due to following reasons:

1. The interaction of socio-economic, institutional, ecological and geophysical processes makes the immediate or long-term consequences of management measures difficult to foresee;
2. Changes in physical conditions (e.g. climate change, urbanization), economic developments, management policy and measurement and modeling technologies, complicate RBM procedures;
3. The lack of a sound scientific approach for the development and implementation of supporting tools reduces the effectiveness of these tools, and consequently, reduces the effectiveness of proposed measures;
4. Objectives or problems may not be clearly defined and correctly translated into different "languages" that can be understood by scientists and end users both;
5. The occurrence of problems at different temporal and spatial scales increases the complexity of IRBM as well as the development of supporting tools.

Therefore, the challenge is to design and develop a supporting tool that is *appropriate* for describing changes in the objective variables. This means that such an instrument should not be overly complex nor excessively coarse, retaining the capability to deal with the difficulties involved in various aspects of system integration.

1.1.2 Decision Support Systems (DSS)

Computer-based models together with their interactive interfaces are typically called Decision Support Systems (DSSs). DSSs are interactive computer-based information providers. The common objective of all DSSs, regardless of the frameworks, methodologies, or techniques used, is to provide timely information that supports human decision makers – at whatever level of decision making (Loucks, 1995).

As mentioned above, integrated RBM should account for the interactions between physical and socio-economic processes, and manage potential conflicts properly. Integrated management requires combination of large volumes of information from a range of sources. A framework is required to couple this information with efficient tools for assessment and evaluation that allow broad, interactive participation in planning and decision making process and effective methods of communicating results to a broader audience. Such a tool is introduced as the computerized system involving various process models to aid decision making in RBM, i.e. a *Decision Support System (DSS)*. It has been widely studied and applied to RBM issues like flood management, ecological assessment, environmental improvement, water supply, water quality control, to name a few issues from the last decade or so (e.g. Reitsma, 1996; Jamieson et al., 1996a, b; Fedra et al., 1996; Welp, 2001; Schielen et al., 2003; Koutsoyiannis et al., 2003; Halls, 2003; Mysiak, et al., 2005).

The technical realization of DSS has changed over the past decades, largely due to the rapidly increasing capabilities of computing power and graphics capabilities. Emerging in the 1970s (Anthony, 1965; Gorry et al., 1971), DSSs have continuously evolved (e.g. Mallach, 1994; Power, 1997; Van Ast et al., 2003; Mysiak, 2005). Regardless of the different forms of DSSs, new issues such as the increasing severity of environmental problems and the growing conflicts in the exploitation of natural resources have added to the challenge to design appropriate DSSs.

With new challenges arising, even more sophisticated DSSs are required in future. The challenges range from (National Research Council, 1999): (i) changes in physical conditions and changes in policy considerations; (ii) growing knowledge on the interactions among river basin components; (iii) better understanding of the feedback among processes operating at different temporal and spatial scales; (iv) the increased availability of river simulation models; and (v) improved understanding of risk and uncertainty in decision-making processes. Current DSS development shows a trend towards integrated systems that can aid decision making and assessment of alternative measures both at a strategic level and at an operational level, viz. at different temporal and spatial scales (Van Ast et al., 2003; Mysiak, 2005).

1.1.2.1 Integration in DSS

Due to the multiple disciplines involved, the key to developing an appropriate DSS is integration. Integration in the context of DSS can be regarded as the *progressive linking and testing of system components to merge their functional and technical characteristics into a comprehensive, interoperable system*. The key question is: how to connect different system components that satisfy functionality and performance requirements with appropriate complexity? This question can be decomposed into three key questions: What are the disciplines involved? How to define system complexity? What are the criteria for evaluating the appropriateness of system structure and system performance? In order to answer these questions, it is necessary to explore the integration difficulties or integration requirements associated with the design of a DSS. Typical integration is needed on two aspects: 1) integration of different processes; 2) integration of different temporal and spatial scales.

In addition to the integration of technologies such as GIS technology, calibration and validation techniques, information management and analysis, *process integration* is essential for the development of a DSS. This concerns the connection of processes involved in the DSS, following their causal relationships. These processes can either be physical process such as hydrology, hydraulics, ecology, or socioeconomic processes including risk assessment. They are normally represented by mathematical models. The integration of these models requires comprehensive insight into their causal relationships.

Figure 1.2 shows an example of a design for an integrated DSS that deals with flood risk and ecological quality assessment as the two objectives.

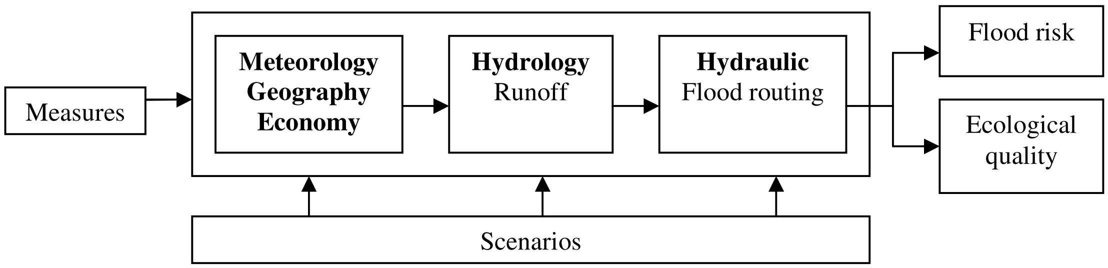

Figure 1.2 *Example of an Integrated DSS for assessing flood risk and ecological quality*

Another form of integration needed in the development of a DSS concerns the combination of different temporal and spatial scales involved at different levels of decision making. In the decision pyramid and information characteristics associated with various levels of decision making, Loucks (1995) categorized decisions related to water resources management in terms of temporal scale, into long-term (development), mid-term and short-term, associated with management level of planning, management and operation, resp. (Figure 1.3). Accordingly, the information characteristics are classified from largely external, multi-disciplinary, aggregated, loosely structured, historical, and often vague, to detailed, real-time, structured, and relatively certain and highly technical. Therefore, most (if not all) of those considerations should be taken into account when a DSS is to be developed.

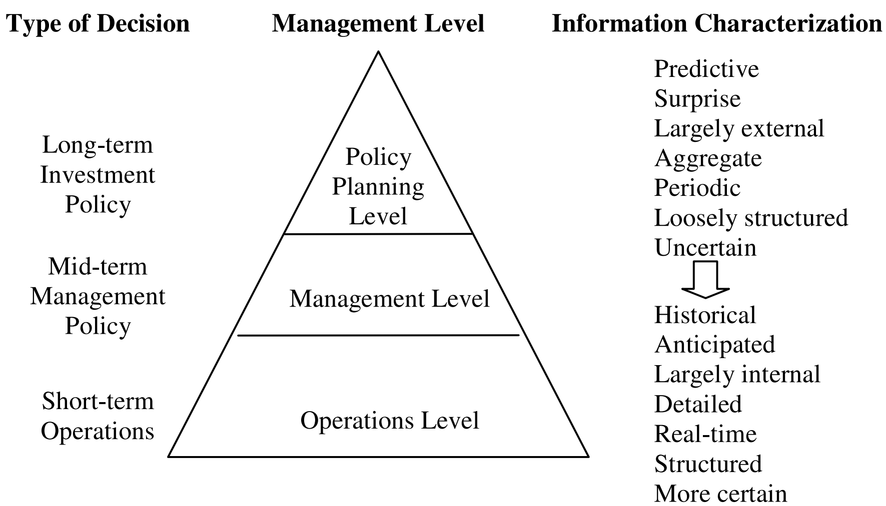

Figure 1.3 *Information characteristics associated with various types of decision (Loucks, 1995)*

1.1.2.2 Design approaches in DSS

To obtain an effective DSS, issues involved in the development of a DSS should be addressed systematically. Several approaches have been proposed for the design, development, implementation and evaluation of a DSS (e.g. Loucks, 1995; Jamieson and Fedra, 1996a; Schielen and Gijsbers, 2003) based on principles from decision theory (Simon, 1960) and system analysis (Forrester, 1962; Miser and Quade, 1985). Depending on the starting points for the design, two different approaches can be distinguished: (a) *user-oriented,* i.e. developing the DSS for a particular problem, e.g. for a specific river, and (b) *knowledge-driven*, i.e. developing a DSS in a generic way to be used for many problems and in many river basins (e.g. Fassio et al., 2004; Mysiak et al., 2005). Both approaches have been found to serve the development of DSSs in a complementary manner.

Regardless of the difference in driving forces, a typical system analysis approach has been commonly applied in the design of a DSS (Miser and Quade, 1985). Based on the physical conditions of the river basin, a system analysis approach begins with problem study, and formulates the system according to certain functionality requirements. The different driving forces however lead to different ways of formulating the problems and of constructing the required models. A user-oriented design aims for a DSS addressing a specific problem for a specific river basin. Therefore the particular problem determines how to define the functionality of the DSS, whereas a knowledge-driven design aims for a generic DSS that can be used for any problem and in any river basin. The knowledge-driven design tends to include sufficient (state of the art) knowledge, while in a user-oriented design a simplification may suffice for rapid development. In future DSS design, both state of the art scientific knowledge as well as end user requirements are to be addressed in a well balanced and appropriate way.

1.1.2.3 Model selection

In addition to the difficulties of how to balance the driving forces for the design of a DSS, model selection is another issue that requires more attention and effort.

Firstly, it has been found that most of the design approaches are unable to answer the question of "*how to select appropriate models so that the system will not become excessively complex or overly coarse?*" Due to the common misunderstanding that higher model complexity may guarantee better performance, modelers may have a preference for more complex models rather than simpler ones. However, as pointed out by Snowling and Kramer (2000), who studied the relationship between model complexity and model performance, higher model complexity does not necessarily lead to better performance when considering uncertainty and sensitivity of the model. The same item has been discussed by De Kok and Wind (2003) who presented a trade-off between model power (to what extent can a model be used to distinguish between proposed measures) and model complexity (model structure in terms of equations and number of parameters). Moreover, the performance of models depends not only on the model itself but also on the data availability and on who is using the model. The modeler's progressive understanding and experience with the model may lead to a better performance. Nevertheless, identification of appropriate model complexity is essential for proper model selection, although it is not the only affecting factor (data availability being equally if not more important).

Secondly, there is a common misunderstanding in model selection: model selection is sometimes confused with the selection of a particular software system (or a modeling package). Nowadays in the design of a DSS, instead of starting from scratch, usually a readily available modeling software package is used. However, it is difficult to choose such a software package. The reasons are: (i) model performance is partly modeler-dependent – so experience with a particular package can be extremely relevant; (ii) in terms of complexity and availability for each process or discipline, there are many modeling software package available, which can provide similar functionality and outcomes. For example, in river modeling, since the mergence of mathematical models for river modeling in the 1970s (Abbott, 1969; Abbott, 1991), hydraulic modeling software packages are no longer a matter of true or false, it is a matter of how good or bad they can perform. For example, there are a number of packages that can be applied to carry out floodplain hydrodynamic computations, such as SOBEK1D2D from WL | Delft Hydraulics (http://www.sobek.nl/), MIKEFLOOD from DHI Water & Environment (http://www.dhi.dk), FLOODWORKS from Wallingford (http://www.wallingfordsoftware.com/products/floodworks/), WAQUA from The National Institute for Coastal and Marine Management (RIKZ) (http://www.netcoast.nl/); all of these are supported by established organizations. Alternatively, there exist many other freeware online, although usually without support or validation evidence. Selection of a modeling software package is not only based on concepts that satisfy the requirements of the to-be-integrated component, but also on other criteria such as purchasing cost, maintenance cost, computational requirements, accessibility of the model. In addition, a tradeoff is needed between the use of freeware and the cost of own development.

In this thesis, model selection refers to the identification of appropriate model complexity; choosing particular modeling software can then be determined together with end users as they are the one who is going to operate and use the DSS in future.

A performance-based comparative approach has been found to be a common method for the identification of model complexity. It is based on choosing the most appropriate model(s) according to prescribed specifications (e.g. Jamieson and Fedra, 1996a; Schielen and Gijsbers, 2003; Mysiak et al., 2005). The appropriate model(s) are selected after carrying out extensive quantitative computations on all of the alternative models. However, due to the dependency between model performance and modeler' experience, a comparative approach cannot guarantee that the selected model is the most appropriate. Thus, there is a need to establish alternative approaches to find the appropriate model without carrying out extensive quantitative comparisons.

1.2 Appropriate Modeling for IRBM

To obtain an effective DSS, several questions should be answered at the design stage: How to deal with changes in physical conditions or preferred policies? How to identify key problems and define objectives clearly? How to combine models at different temporal and spatial scales? How to select models that are neither overly coarse nor excessively complex? How does uncertainty propagate through the DSS and how can it aid decision making? How to evaluate the performance of a DSS? To answer these questions, a thorough understanding of all components involved in the development of a DSS is necessary, and strict scientific principles should be followed, as explained below.

1.2.1 The role of scientific principles

Scientific principles are the backbone of any DSS dealing with physical processes. In this thesis scientific principles are defined as the *level of understanding the physical processes involved in river basin management and their causal relationships*. As already pointed out in many studies, scientific principles have not always received the attention deserved (Mills and Clark, 2001; Farrell et al., 2001; De Kok and Wind, 2003; Lankford et al., 2004).

The role of research scientists has been highlighted by Mills and Clark (2001), who proposed research scientists to adhere to scientific principles in order to provide confidence and a sound basis and solid interface to end users and decision makers. The importance of science has been addressed also by Farrell et al., (2001) who concluded that "*one of the most important parts of this interface concerns quantitative modeling efforts that must be designed to answer the questions that decision-makers pose and at the same time satisfy demands on rigor and fundamental basis from the scientific community*". Although scientific principles by themselves do not make decisions or guarantee a correct decision to be made, it helps "*better inform difficult natural resource decisions in several ways*" (Mills and Clark, 2001).

The same has been argued by De Kok and Wind (2003). Based on a comparative study of six different DSSs in relation to different aspects of the design, development and application of a DSS, De Kok and Wind concluded that essential conditions for a DSS are: (i) a solid analysis of the problem from an integrated point of view involving end users; (ii) a clear objective statement; (iii) an acceptable and understandable presentation format, and (iv) a flexible design which can deal with changing demands. In addition, they found that even if those conditions are met, effective IRBM based on the DSS is not guaranteed. The reason being that the lack of applying scientific principles could easily lead to selecting models and data of unnecessarily high complexity, as well as hamper the development of a distinguishable method to present end users with clear differences between alternative measures. Lankford et al., (2004) also pointed out that insufficient scientific involvement in the decision making process could prevent the effective implementation of a DSS. Clearly, scientific principles require adequate attention and its importance in the design of a DSS should be emphasized.

However, this requirement is in general quite difficult to satisfy. Scientists or modelers are often specialists with in-depth knowledge of their own field. Such a *vertical-sufficiency* cannot guarantee a *horizontal-sufficiency* which requires in-depth knowledge on every/all processes involved, as well as on their inter-connections. This requires rich knowledge and experience of the modeler, which might be not as good as their own specialist background. In addition, there is a gap between science and practice due to inadequate communication between scientists and end users (Jeffrey and Gearey, 2003). Thus, effort is needed to bridge the gap to implement sufficient scientific principles.

1.2.2 Appropriate model complexity

To identify appropriate model complexity, there are two issues that need to be clarified: (i) how to define model complexity; (ii) at what level can model complexity be considered 'appropriate'. The following sections review recent research related to model complexity and the relationship between model complexity and model performance.

1.2.2.1 What is model complexity

Complexity can be regarded as the combination of distinction and connection, which is one of the most important properties of a model. However, a unique definition (or measure) of model complexity is lacking. Wagenet and Rao (1990) categorized models into three basic groups based on their complexity as: (i) research models (more complex), (ii) management models (less complex), and (iii) screening models (analytical solutions used only for relative comparisons). Each of these types of models was developed for a different purpose, and had inherent properties, assumptions and limitations underlying its development. Some studies measured model complexity proportional to the number of parameters (Gan, et al., 1997; Elert et al., 1999; Perrin et al., 2001; Green, et al., 2002). That is, the larger the number of parameters, the higher the complexity. However, as Snowling and Kramer (2000) pointed out, the complexity of a model depends on not only the number of parameters but also on its structure and level of details.

Therefore, the number of parameters and state variables, the sophistication of the mathematical relationships, the numerical schemes to solve the equations, and the overall number of processes contained in the model, should all be taken into consideration when model complexity is to be defined.

1.2.2.2 At what level can model complexity be considered 'appropriate'

There is a misunderstanding that model performance is proportional to its level of complexity in terms of mathematical expressions and model structure. In many DSS developments the design is largely based on data availability and model accessibility. Not often the question is asked "*what level of complexity should a model have?*".

Firstly, higher complexity does not guarantee better model performance. For example, higher complexity in general requires a larger amount of data for calibration and verification, which may increase the influence of uncertainty in the data and hence reduce the effectiveness of the model with higher complexity. Moreover, a complex model in general costs more in terms of data demand and computational load. For example, the transition from a 1D hydraulic (river) model to a 2D hydraulic (river plus floodplain) model increases the computing time exponentially, which might not satisfy available time criteria when the model is to be used for rapid assessment. The problem then becomes "how to find the balance between model complexity and performance requirement of the DSS?" This type of problem has been addressed in the 14th century by the philosopher William of Ockham (1287 – 1324), with the famous Ockham razor "*Pluralitas non est ponenda sine necessitate*", which can be translated as "*entities should not be multiplied unnecessarily*". In many cases this is interpreted as "keep it simple". However, in practice it is difficult to achieve precisely the desired appropriate model complexity. As already mentioned before, the performance of a model is not only a matter of model complexity, but also depends on the knowledge and experience of the modeler.

Secondly, it is difficult to identify a single relationship between model performance and complexity. De Kok and Wind (2002) assumed that model complexity is related to its performance, indicated as *model power* (*y* axis in Figure 1.4). In their appropriate schematization curve, model power is first assumed to increase proportionally with model complexity; however, when higher data demands and other uncertainties involved with higher complexity models become dominant, model power may reduce. When the balance is found between desired model power, e.g. the *Minimum level for decision* on the *y* axis, and model complexity to distinguish alternative measures, the 'appropriate model' can be identified, indicated as *model a* (or *model b*).

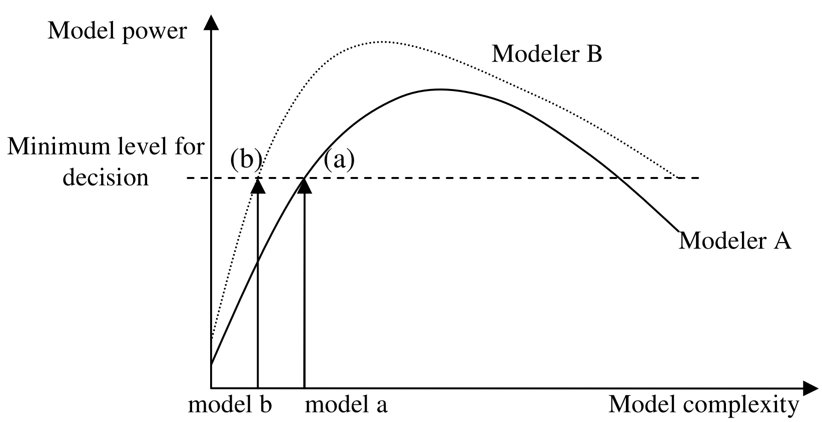

Figure 1.4 *Modified conceptual schematization of relationship between model power and complexity (after De Kok and Wind, 2003)*

However, such a relationship is modeler-dependent and cannot be generic. As indicated in Figure 1.4, compared to an inexperienced modeler (*Modeler A*) an experienced one (*Modeler B*) may obtain better results from a model, making model *b* more appropriate than model a. Moreover, it is difficult to define the minimum required *model power*. It can be a single criterion such as model accuracy, or most often a combination of multiple criteria such as model accuracy, model cost, data demand, interface flexibility, presentation methods, and the critical needs of adequate runtime performance of the DSS in case of emergency (Aerts et al., 2000; Jain and Singh, 2003). These difficulties make the selection of a model based solely on a schematization curve as shown in Figure 1.4, impractical.

Nevertheless, identifying the appropriate model complexity is essential for the development of a DSS. Hence continued efforts are needed to establish appropriate model complexity.

1.2.3 Incorporating uncertainty analysis

Once a DSS is built, the next step is to evaluate its performance, either quantitatively or qualitatively. This requires an overall presentation of the outcomes of the DSS. Some attempts have been made to evaluate the performance of a DSS (Reitsma, 1996; Finlay and Wilson, 1997; Poon and Wagner, 2001). According to Potts et al. (2001), the generic problem of DSS evaluation is the lack of a quality definition and methods to assess this quality. Finlay and Wilson (Finlay and Wilson, 1997) refer to about 50 overlapping, ambiguous validity concepts for measuring quality, in the literature, which all differ in their chosen approach and features of the DSS addressed. Despite this ambiguity of validity concepts developed so far, some of the success factors, also known as critical success factors (Poon and Wagner, 2001), are commonly agreed upon. Among various approaches, Uncertainty Analysis (UA) is found to be important for evaluating the system performance (Snowling and Kramer, 2000), as well as for distinguishing the impact of alternative measures that are to be simulated in the DSS (De Kok and Wind, 2003). Difficulties involved in UA are typically the identification of uncertainty sources and their contribution, as well as their propagation through the integrated system.

In general a DSS involves various scientific disciplines such as hydrology, hydraulics, ecology, water quality, flood risk assessment, navigability and so on. Some of the processes and modeling techniques have received much more comprehensive studies than others. For example, hydrology can be considered a mature science as it has been studied more than half century, and has moved from lumped modeling towards GIS based distributed modeling (Singhroy, 1995; Aerts et al., 2000; Morari et al., 2004). Flood risk assessment, however, is not as mature as hydrology. The core of the models underlying conventional risk assessment – the relationship between inundation depth and damage – is usually case-specific, and there is no report about uncertainty distributions of the damage coefficients.

In addition to the difficulty of defining and quantifying uncertainty sources, uncertainty propagation through an integrated system remains rarely reported. UA has been primarily been carried out for individual models (e.g. Crosetto et al., 2001; Hanna et al., 2001). Thus, effort is needed to study uncertainty propagation through a DSS for IRBM.

1.3 Flood Risk Assessment

Flood risk assessment has gained increasing attention due to its importance in flood management. In history, approaches to deal with flood risk have changed over time: from simple adaptation such as building houses on a higher level or using boats to take people away from the risk area, to engineering approaches such as the construction of dikes to deal with flood risk by confining the river capacity (e.g. White and Haas, 1975; Thampapillai and Musgrave, 1985). Quite recently, flood management measures have shifted to alternative measures such as the use of retention basin (deliberate flooding in one particular area to reduce the risk for downstream areas (Changnon, 1985; Edward and Thomas, 1993; Penning-Rowsell and Fordham, 1994; Vis et al., 2003; Dijkman, et al., 2003; Nishat, 2003). Integrated flood management is aiming at mitigating flood risk by combining various types of flood management measures (Green et al., 2000).

1.3.1 Flood risk assessment approaches

To achieve effective flood management, it is necessary to constantly re-assess the management approach adopted and to transfer knowledge from one country/river to another. However, very often rather than knowledge transfer taking place, the approach developed in the context of one country/river has simply been transposed to another country/river, which might be inappropriate due to the difference in nature of two cases. Therefore, to obtain insight into the appropriateness of proposed measures, risk assessment is essential.

However, such an *Integrated Food Risk Assessment (IFRA)* that serves both long-term planning and short-term operation requires functioning at different spatial and temporal scales, and is often still lacking. The concept of flood risk consists of two aspects: (i) establish the probability of flooding and (ii) assessing the consequence, viz. estimating the flood damage. FRA used to be based on statistical risk-analysis approaches which focus on the study of flood defense systems using probabilistic design (CUR, 1990; Stedinger, 1997; Vrijling et al., 1998). Such methods provide risk assessment to support long-term planning such as the construction of a dike (CUR, 1990), and is usually applied for large spatial scales (size of several hundreds kilometers). This approach does not usually involve complex hydraulic computations in more than one dimension. More recently, various FRA methodologies have been developed by simulating the consequence using physically-based numerical simulation models for inundation modeling, which calculates the damage caused by individual flood events (Horritt and Bates, 2002; Van der Sande et al., 2003). This method provides the consequence – the damage and change in physical conditions – associated with a particular flood event. Often such a method is used for short-term operation such as the operation of a retention basin during flooding. Integration of different FRA approaches is essential for achieving sustainable flood management which serves both short-term and long-term planning, involving different temporal and spatial scales in IRBM.

1.3.2 Velocity effects

It is known that flood damage is not only caused by inundation depth, but also by other parameters such as inundation duration, flow velocity, wind direction and wind force. Apart from non-hydraulic factors there are effects of pollutants and sediment transport, or anticipatory behavior of people and organizations in risk areas. However, important factors such as the flow velocity which may cause additional damage, particularly in steep areas, are rarely included in FRA studies. If a river flows very fast, it could wash away buildings and property and even drown people. Such effect was observed in a recent flood in Cornwall, England showing the significance of damage caused by flow velocity rather than inundation depth. On August 16, 2004, a 75 mm rainstorm of two hours caused two nearby rivers to flow through the village of Boscastle in South-West England, the rapid flow sweeping away buildings and cars out into the open sea (BBC Cornwall, 2004). At successively lower discharges corresponding to lower water velocities and shallower water depths, the flooding becomes relatively safer - at low depth it may not be life endangering, but it could still result in considerable damage to property. This is also true for other parameters, such as the inundation duration which is significant for agricultural production. However, it has been found difficult to quantify such important parameters.

Several studies have been carried out to quantify the relationship between flow velocity and damage (Stephenson, 2002; Roos, 2003; Asselman and Jonkman, 2003; Kelman and Spence, 2004). To obtain a quantitative relation between velocity and flood damage, Roos (2003) studied the collapse of buildings caused by velocity. He derived combinations of water depth and flow velocity for which a wall of a certain building type will collapse with 100% probability. Based on the relationship between water depth and velocity, Asselman and Jonkman (2003) developed a damage model that simulates the loss of life, assuming the collapse of a building in rapidly flowing water will result in the death of all those present inside the building. This is of significant value for obtaining quantitative expression but is limited to buildings only, and cannot be applied to other types of land use such as crops and traffic.

A different quantitative expression for the effect of velocity on flood damage was recently proposed by Kelman and Spence (2004). The method is based on knowledge of the underlying hydrostatic and hydrodynamic processes, to derive a value for economic loss. However, this method remains purely illustrative and qualitative, and needs to be validated before it can be used. Due to the limited relationships that have been investigated (mainly on the buildings), it is meaningful to seek a generic method which can take damage effects due to velocity into account.

Among various flood damage evaluation methods, the *risk-matrix* concept is considered of interest (Stephenson, 2002; Fattorelli et al., 2003; Vrouwenvelder et al., 2003). This method combines depth-caused damage with effects due to flow velocity into an index of risk levels using a predefined classification of velocity effects and inundation depth. Limitations of those methods are the classification of the risk index that is lacking objective quantification of velocity effects on properties (Du Plessis, 2000; Adriaans, 2001). The potential damage is quantified in terms of four classes associated with four different combinations of land use (Fattorelli et al., 2003).

In summary, quantitative expressions for the effects of flow velocity on damage/risk need to be established in order to be able to achieve proper flood risk assessment. However, this is difficult because:

1. Few measurement data are available for floodplain areas during flooding;
2. Flow velocities at floodplain areas can only be predicted using 2-dimensional hydrodynamic models, which are not always available in conventional FRA approaches;
3. Only qualitative understanding is presently available of flow effects on damage;
4. There are very few references in the literature on IFRA.

1.3.3 Rapid flood risk assessment – the capabilities of GIS technology

Many flood management practices require rapid FRA to provide support for decision making within a short time period, e.g. in 1-2 hours. However, the relatively large computation time needed for carrying out a 2-dimensional hydraulic inundation modeling computation limits to some extent its practical application. Clearly, advanced hydroinformatics techniques can be used to emulate pre-computed scenario's into an Artificial Neural Network configuration (Mynett et al., 2004b), or parallel computing algorithms can be used to speed up computational performance (Mynett, 2004c), but this is not yet common practice in current FRA approaches.

Attempts have been made to reduce the large computational loads of a fully 2D model. For example, instead of solving the complete 2D hydrodynamic equations, Bates and De Roo (2000) simplified the equations into a 1D kinematics routing in the river channel, and a mass-balanced quasi-2D approach in the floodplain. This, however, still requires considerable computation time due to the use of raster data of Digital Elevation Model (DEM). Further approximations are needed to shorten computing time without losing the functionality and reliability of FRA, and without carrying out complex hydrodynamic computations. An attempt is made in the latter part of this thesis to obtain a rapid FRA tool – including the effect of velocity – using present day capabilities of GIS technology.

1.3.4 Uncertainty analysis in FRA

Previous studies show that UA plays an important role in decision making for flood and risk management (Pivot et al., 2002; Plate, 2002; Zerge et al., 2002; Chauhan and Bowles, 2003; Van Manen and Brinkhuis, 2004). The general steps involved in UA of FRA include *identification* of uncertainty sources and *propagating* of uncertainties from the data to the different component (disciplinary) models. These studies, however, do not present uncertainty distributions for integrated FRA. For example, in their study on uncertainty sources in flood damage assessment, De Blois and Wind (1996) identified the most important uncertainty sources as (in sequence of significance): (i) river dike height (for rivers with dike), (ii) river discharge including its frequency of occurrence, (iii) damage estimates, and (iv) risk of dike breach. The inundation models used in their case studies are simplified one-dimensional hydraulic models, which left out some important parameters such as the growth of breach width. Chauhan and Bowles (2003) presented their UA on dam safety risk assessment, including an approach to incorporate input uncertainty into the risk analysis model. Their work shows the significant benefit of including UA in the decision making process related to dam management. The study focuses on statistical calculations, i.e. dam failure, without considering other variables such as geographical conditions (DEM) or land use conditions. An integrated view which involves multiple disciplines and allows for different temporal and spatial scales does not seem to be readily available at present.

Some studies tried to look at uncertainty propagation through the integrated system (e.g. Apel et al., 2004). However, in order to cope with the high CPU-time demand when carrying out multiple runs for randomly varied parameters, Apel et al. developed a stochastic flood risk model using simplified models associated with the processes included in the chain. The results provide an indication of the level of uncertainty, but such conclusions might be limited by the simplifications of models involved; when using more complex models the distribution of uncertainties is likely to be different.

In summary, most studies carried out were on the analysis of specific parameters or based on simple risk models. Hardly any uncertainty analyses seem to have been reported for an IFRA, which is one of the study issues in this thesis.

1.4 Research Scope

The research described in this thesis covers two dimensions: (i) a general dimension that deals with the difficulties related to the selection of hydraulic models for FRA as well as for evaluating DSS performance; and (ii) the specific improvement of current FRA approaches by incorporating the effects of flow velocity. The latter is investigated using a fully 1D2D hydrodynamic approach based on the SOBEK1D2D modeling system of WL | Delft Hydraulics; in a later stage of the research some effort is devoted to exploring the capabilities of present day GIS technology for obtaining a first approximation of velocity effects.

1.4.1 Research objectives

The general research objectives can be formulated as follows:

1. To provide a general framework for flood risk assessment, including the problem of model selection;
2. To improve current flood risk assessment approach by including effects of flow velocity;
3. To explore the possibility of developing a rapid risk assessment tool based on GIS technology.

To achieve these objectives, the principle of *appropriate modeling* is applied. This means that an appropriate level of model complexity is established that is neither excessively complex nor overly simple, but capable of distinguishing between the various alternative measures that are being proposed. A central aspect of the research is how to select appropriate hydraulic models for different FRA purposes, including how to support decision making when uncertainty is involved.

1.4.2 Research questions

In accordance with the general objectives described above, the research questions addressed in this thesis are all related to IFRA and can be formulated as:

1. How can the effect of flow velocity on flood risk be incorporated in IFRA?
2. How can uncertainty analysis be applied to IFRA?
3. Can GIS technology provide a useful approximation for obtaining a rapid IFRA?

1.5 Thesis Outline

This dissertation consists of four major parts contained in the next four chapters; the answers to the research questions formulated above in Chapter 1 are summarized in Chapter 6, together with conclusions and recommendations.

Chapter 2 describes the history over the past decades of DSS design for IRBM in general and for FRA in particular. The conceptual differences in current DSS design approaches, which take either the user requirements or the available models as a starting point for the design, are reviewed. General issues on how to select appropriate models, how to assess DSS performance, how to use UA and how to enhance the role of scientific principles, are discussed. These issues are also discussed related to the design of an IFRA system: how to select appropriate hydraulic models, how can uncertainty analysis benefit IFRA, and how to incorporate the effect of flow velocity into IFRA. A general framework for the design a DSS is suggested, consisting of two phases, qualitative analysis and quantitative analysis, is illustrated in this chapter. The methodology of *double-direction searching* for appropriate model selection and for performing uncertainty analysis in integrated systems, is also introduced in Chapter 2.

Chapter 3 presents a case study on developing an integrated FRA modeling system for the river Elbe (Germany) in some detail. Natural conditions including hydrological properties, geographical conditions and flood defense systems of the Elbe River are described. Relevant issues such as its flooding history, present day objectives of the authorities and potential measures to reduce flood risk are also described. Based on two different approaches for FRA, i.e. a statistical approach and a physically-based approach, a conceptual framework for IFRA is formulated, building on the methodology proposed in chapter 2 for model selection. This framework will be used for FRA of the Elbe River in the consecutive chapters.

Chapter 4 presents a quantitative study on IFRA at a local scale, where flood risk/damage is given at the immediately affected location. Two types of risk models are applied to study the flood risk distribution at the study area, viz. near the German town of Sandau. Serving different risk assessment approaches, two types of hydraulic models are incorporated, namely the steady state model HEC6 to determine the stage ~ discharge relationships, and the hydrodynamic model SOBEK1D2D to actually simulate the flooding processes. To include effects of additional damage caused by large flow velocities, the risk matrix concept originally developed by Fattorelli et al. (2003) has been modified and applied in this case study. Quantitative analyses were carried out on dike break effects using UA. The study shows that one of the new features developed within the context of this thesis – IFRA including effects of additional damage caused by flow velocities – proves very useful for flood management purposes.

In Chapter 5, another case study for the Elbe River is carried out by simulating the effects of dike break for risk mitigation. The developed IFRA framework and models are applied to examine how flood mitigation can benefit from an intentional dike break measure. The simulation results show that IFRA can be used to aid risk mitigation using comprehensive risk indicators. To obtain a rapid assessment tool at reduced computing time, an attempt has been made to approximate the inundation results using GIS technology only, based on the concept of storage functions for sub-basins. The result shows a satisfactory overall agreement although locally there exist large difference between the GIS approximation and the actually computed flow velocities. Thus, although a GIS approximation might be used to provide a first indication of effects of flow velocity (momentum effects), more accurate results still rely on fully 2D hydrodynamic computations.

Chapter 6 summarizes the key actives of this research and provides the answers to the research questions posed in Chapter 1. Conclusions are drawn and recommendations are given on appropriate modeling for IFRA, in particular on model selection and the role of uncertainty analysis.

Chapter 2

Tools and Methodologies for DSS Design and Flood Risk Assessment: the Role of Appropriate Modeling

There is a growing perception that integrated river basin management depends on a thorough understanding of the interaction between the physical, socio-economic, and ecological processes. The complexity of the problem is increased by the presence of multiple stakeholders with different interests, multiple objectives that are sometimes in conflict, uncertain future conditions, changing policy preferences, and the interaction among processes working at different spatial and temporal scales. This calls for Decision-Support Systems (DSSs) that can assist river managers in the formulation of integrated management strategies and the exploration of different scenarios while taking into account stakeholders' interests.

Depending on how problems are addressed and models formulated, two different directions can be discerned in the literature on DSS design. The *user-oriented* approach aims at addressing a problem for a particular river or river basin; it takes end-user requirements and functional criteria as starting point. The *knowledge-driven* approach aims to develop a generic DSS that can be applied to arbitrary river basins; it uses scientific knowledge as starting point for the design. Examples of both approaches are discussed in this Chapter, including a number of typical problems such as the involvement of end users during the design, changing physical, political and socio-economic conditions, the model selection problem, and how to evaluate IDSS performance.

The purpose of Flood Risk Assessment (FRA) is to support decision-making for flood management by taking into account the objectives of multiple stakeholders and various temporal and spatial scales. In general, a distinction can be made between *statistical* and *physically-based* FRA. In the statistical approach, both the effect and probability of a flood are taken into account, whereas in the physically-based approach the flood risk is assessed by determining the direct and indirect effects for a specific flood event. Among the issues discussed in this Chapter are the lack of generically applicable depth ~ damage functions, as well as the problem of how to incorporate the effect of flow velocity in the risk assessment.

The Chapter concludes with a review of uncertainty analysis techniques that can be applied to select appropriate models for FRA.

2.1 Introduction

Most of the DSSs are designed according to end users' requirements to solve certain problems encountered in a specific river. This can be regarded as the *user-oriented* design (Loucks, 1995; Jamieson and Fedra, 1996a; Schielen and Gijsbers, 2003). In contrast, the *knowledge-driven* design aims for a generic DSS that can be used for any problem related to any river basin (Fassio et al, 2004; Mysiak et al., 2005). In the user-oriented approach, emphasis is put on the problem definition as well as the identification of alternative measures, searching for models based on the problem requirement. In the knowledge-driven approach, the DSS is developed on the basis of possible problems that may arise in any river basin, the emphasis being laid on the development of models mainly based on the state-of-the-art available knowledge.

Both types of approach follow the sequence of steps from system analysis: problem definition → system development → system evaluation.

A system analysis to support river basin management (RBM) has a simple aim: to provide (preferably quantitative) information to decision-makers to enable the best selection from alternative measures. Within the context of DSSs, the analysis should meet the following requirements (Dijkman and Klomp, 1990):

- The analysis should take the user as the starting point and should provide accurate and useful information for decision-making.
- An intensive communication between the analysts (modelers) and the decision-makers is required. This interaction should ensure that the analysis focuses on the important problems, as perceived by the decision-makers, and produces the type of information desired by the decision-makers.
- Besides decision-makers, implementing agencies and interest groups of stakeholders should also be involved in the analysis to promote the acceptance of the results and support for the DSS.
- The analysis should be sufficiently flexible: it should allow examination of alternative strategies, determination of the relevant socioeconomic and environmental impact, and quantify uncertain developments.
- Due to the continuous development of RBM, the analysis should be done in such a way that applied techniques fit into a consistent framework for analysis. Such a framework can be gradually extended and improved, and may be continually used to provide information for decision-making.

A system analysis approach is generally adopted to meet these requirements (Simon, 1960; Forrester, 1962; Miser and Quade, 1985, 1995; Nieuwkamer, 1995). By incorporating a computational framework for analysis, this approach allows the integration of contributions from various disciplines such as hydrology, hydraulics, and ecology. The same was concluded by De Kok and Wind (2002) who proposed a structured problem-based strategy, i.e. a system analysis approach, to be followed for the development of a DSS. Such an approach is needed when there is a problem – or more often, a politician's or stakeholder's awareness of a problem and the willingness to solve it – but no clear idea of what is wrong or how it might be corrected.

Figure 2.1 illustrates the key processes involved in the approach of system analysis. It consists of three major phases, the *formulation phase*, the *research phase*, and the *evaluation and presentation phase* (Miser and Quade, 1985).

In the *formulation phase*, problems are identified as clearly as possible. It usually takes not only extensive communication between modelers and end users for what to do about the situation, but also a great deal of discipline by both sides. It also requires inquiry into, and agreement on, the goal to be aimed at, and constraints and limitations of possible policies and courses of action. The modelers should understand the underlying situation and have enough background knowledge to identify the main underlying problems.

The next step is to identify key variables and possible actions or alternatives that appear to offer some hope of improving the situation, to collect data and transform those data into relevant information. This analysis forms the *research phase*. At this stage, modelers forecast the system environment or context in which each alternative is assumed to be implemented, and examine each alternative based on political, economic and other boundary conditions. This analysis will provide an inventory of tentative alternatives. Another important action that should be taken in this stage is formulating and constructing models. Those models should be ready for tuning the information about the alternatives into evidence for comparing and ranking them with respect to cost, effectiveness, and other relevant measures of the consequences that would aid decision-making. This issue is being addressed in this thesis (see Chapter3 Section 3.3).

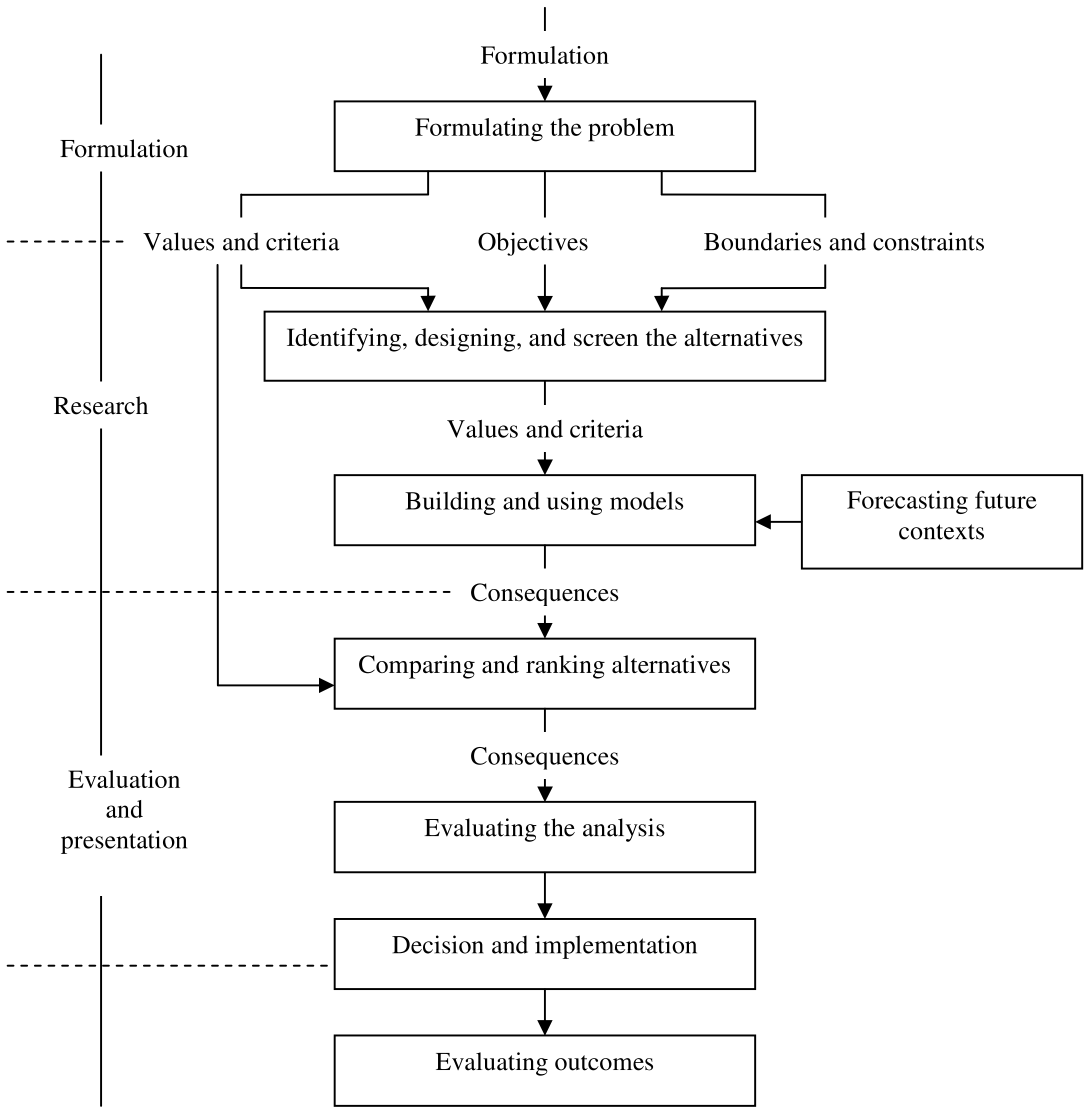

Figure 2.1 *Principal activities in system analysis processes (Miser and Quade, 1985)*

In the *final stage*, evaluation and presentation of the analysis results are carried out. The alternative implementations are converted into means such as cost-benefit, based on which recommendations can be made. In general, a comparison is made between different situations where alternative implementations are feasible. The last step is evaluating the outcomes of the implementation effort, to see if it is achieving the desired result. As pointed out by Miser and Quade (1985), this step is not always carried out.

The system analysis approach has been widely applied in the development of DSSs (Miser and Quade, 1995), such as the icebreaking operations in the Northern Baltic (Jennergren et al., 1995), managing eutrophication in Lake Balaton (Somlyody, 1995), and the planning of The Netherlands' water resources (Goeller et al., 1995). In integrated river basin management, a wide variety of system analysis approaches is applied for the design of DSSs, as presented in the following section.

2.2 State of the Art of DSS Design

The state of the art of DSS design consists of the principles of the frameworks that have been established in several previous studies – in particular the model selection approach –as well as efforts that have been made to deal with issues involved in the development of a DSS, such as the role of knowledge, policy analysis and importance of end-user's participation. These issues are presented in more detail in the following sections.

2.2.1 Different DSS design approaches

The literature describes various approaches for the design and development of the architecture of a DSS (e.g. Simon, 1960; Loucks, 1995; Jamieson and Fedra, 1996a; Poon and Wagner, 2001; Schielen and Gijsbers, 2003; Mysiak et al., 2005). This Chapter discusses the following typical examples: the multi-task approach (Loucks, 1995), the open-design approach (Jamieson and Fedra, 1996a), the functionality-based approach (Schielen and Gijsbers, 2003), the decision-theory-based approach (Fassio, 2004), and the appropriate-modeling approach (De Kok and Wind, 2002, 2003). These approaches are in general user-oriented, either explicitly or implicitly, and can be considered an early version of appropriate modeling.

2.2.1.1 Multi-task approach (Loucks, 1995)

Loucks (1995) proposed a *multi-task* approach for DSS development (Figure 2.2). The approach starts with end users where the problem and objectives are based on a so-called mental model (a conceptual framework without firm structure or style) provided by analysts (or modelers). With the help of such a mental model and supported by interactive communication between modelers and end users, agreement about the purpose of the DSS and its issues, objective and information needs, is reached. Once such an agreement is obtained, the design phase begins. The mental model is transformed and translated into a 'formal' model, which is identified and mathematically formulated, and is coded in computer programmes with user interfaces - this is the computer programming stage. Testing of the DSS requires calibration and verification, once the programming is done. Usually, there are changes in problems or changes in physical/political conditions that make a modification of the DSS desirable. This has to be carried out on the basis of interaction between modelers and decision-makers, including training of the end users. For the implementation stage, Loucks suggested a training programme for the end users to ensure sufficient knowledge for an effective use of the developed DSS.

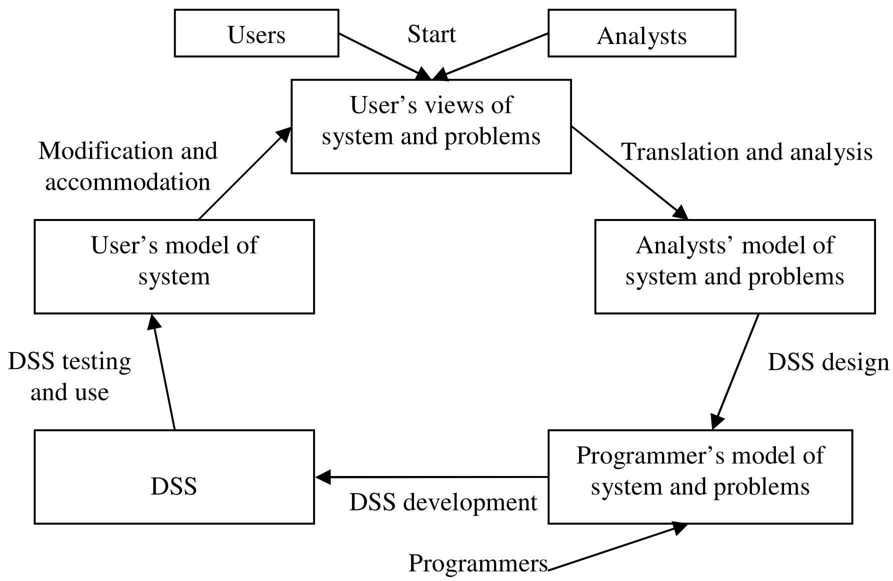

Figure 2.2 *Processes and transitions involved in DSS development (Loucks, 1995)*

This approach is rather qualitative and general. It covers the identification of the most important processes involved in the development of a DSS, such as problem definition, design of the conceptual DSS framework (the mental model), communication with the end users and education of the end users to ensure sufficient relevant knowledge, model structure identification, model calibration and verification, via a user-friendly interface. This multi-task approach provides the basic steps for the development of a DSS. However, issues that have received more recent attention are not incorporated. For example, UA is missing; how to obtain an appropriate model with sufficient complexity but not more than that, has not been addressed; the evaluation of the DSS is focused on the model calibration and validation, which is insufficient due to the conflicting interests involved; also, dealing with different temporal and spatial scales is hardly mentioned.

In this approach, model selection is based on a testing approach, that is, to include models based on an '*action-oriented description of the functional and data architecture of the system. Such a list can include physical as well as abstract objects and the associations among them. These objects can then be included in the DSS-use scenarios to see if they are defined and described satisfactorily*' (Loucks, 1995). In summary, models are selected based on their performance through comparison.

2.2.1.2 The functionality-based approach (Schielen and Gijsbers, 2003)

Focusing on the difficulties encountered during system development with respect to the changes in social opinion on the nature of measures to be assessed, changes in information technology, and different views of various end users and organizations, Schielen and Gijsbers proposed a development process that rests on functionality requirements (Schielen and Gijsbers, 2003). The requirement of the DSS is that it should be based on a GIS environment, enable a large-scale (several kilometers, 1D hydraulic computation) analysis of flood management problems and detailed single floodplain modeling (several hundred meters, 2D hydraulic computation), show model outcomes preferably directly in GIS format, enable comparison of different cases in the form of a table with relevant effects, and have solid functionality to trace, reproduce and access results. This wide range of desired functionalities was available in existing instruments, but not integrated in one system for both 1D and 2D hydraulic computations for rivers. Therefore, efforts were mainly given to the integration of the different components that were available before the development of the DSS.

To meet the functionality requirement, a workflow support approach consisting of four steps – namely to explore, define, compute and analyze – was developed by Schielen and Gijsbers, and applied to develop the desired system.

The 'explore' option in the DSS Large Rivers enables users to explore the basic topographic data, as well as the status of the computational system, and share the documents among different types of users during the various stages of design and assessment. The design stage is entirely focused on the translation of the ideas of the river manager into modeling language, following strict formalized rules. The process model computation is automatically launched, once a measure is selected. Those measure and model relationships are internally connected underneath the DSS interface. After the computations have been made, results are presented through either a map of model outcomes or a quick view of indicators. To support the process of selecting the most appropriate measure, multi-criteria analysis methods can be invoked.

As in the multi-task approach, the functionality-based approach presents the development of the DSS through the architecture of the DSS. The philosophy of the development is embedded in a workflow structure, in which each of the steps translates the functionality requirements into technical procedures. However, due to the different perceptions of scientists (or modelers) on the one hand and end users (or decision-makers) on the other hand, and due to the difficulty of obtaining an acceptable uncertainty presentation or consideration agreement, uncertainty is not accounted for in the functionality-based approach. Also, the DSS Large Rivers was developed on the basis of the available models, particularly the 1D and 2D hydraulic models. This circumvented the problem of model selection while the conflict between model complexity and model uncertainty was not taken into consideration.

2.2.1.3 DPSIR – an approach based on decision theory (Fassio et al., 2004)

Based on the decision process proposed by Simon (1960), a three-phase framework which was suggested by the European Environment Agency (EEA, 1999), was adopted by Fassio et al. (2004). As illustrated in Figure 2.3, the *decision theory based* approach – also named DPSIR – is a framework of environmental cause-effect relationships. *D* represents the Driving forces; *P* is the Pressures on the environment caused by human activities; *S* denotes the State of the environment; *I* indicates the pressures' impact on the environment; *R* stands for the human activities and desirable social Responses. This DPSIR chain provides end users with an integrated view of complex, interacting issues. Other DSS developers (e.g. Jeunesse et al., 2003) have adopted this method as well. This method consists of three phases, namely the conceptual phase, the design phase, and the selection phase.

In the *conceptual phase* of DPSIR-based design, the DSS users conceptualize the structure and communicate the decision situation according to the cause-effect relationships underlying the environmental problem(s). During this phase, problems are studied and an indicator for each process is defined.

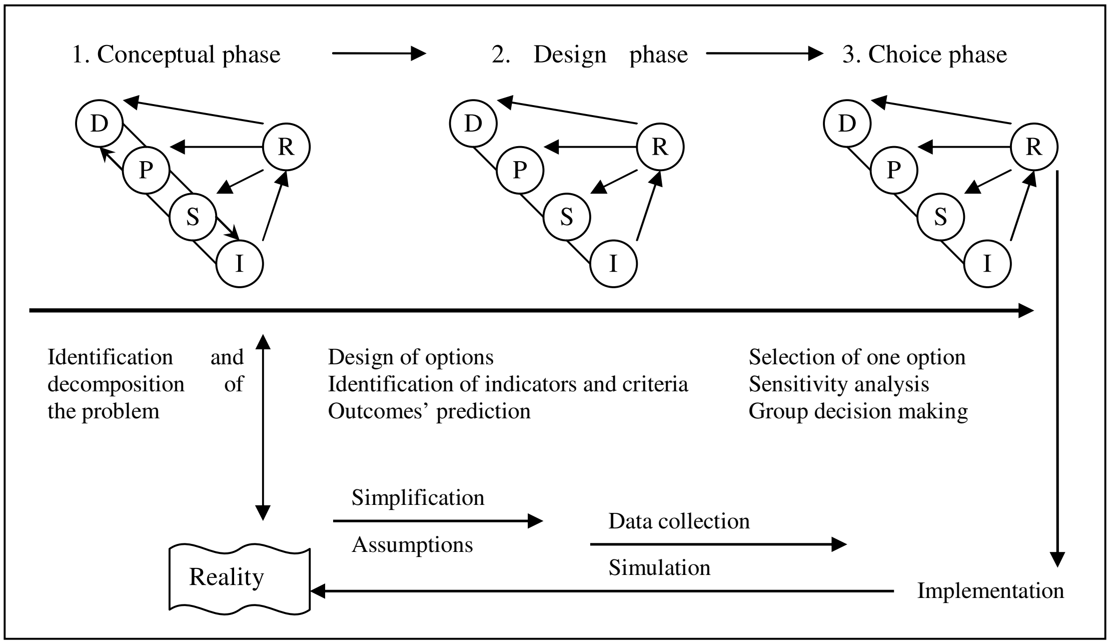

Figure 2.3 *Conceptual framework of the mDSS software, in accordance with the DPSIR approach (EEA, 1999) and decision theory proposed by Simon (1960) (Fassio et al., 2004)*

In the *design phase*, the possible options of policy and measures are set out. Process models are formulated. Criteria for evaluating the performance of these options are identified on the basis of available indicators describing the causal links between driving forces, pressures and changes in state (D-P-S-I-R chains). Indicators in the DPSIR chains include the following:

- *Driving* forces describe the social, demographic and economic developments in societies and the corresponding changes in life styles, overall levels of consumption growth and development in the needs and activities of individuals.
- The resulting environmental *Pressures* describe developments in release of substances (emissions), physical and biological agents, the use of resources and the use of land.
- The *State* of the environment gives a description of the quantity and quality of physical phenomena (such as temperature), biological phenomena (such as fish stocks) and chemical phenomena (such as atmospheric CO_2 concentrations) in a certain area.
- *Impact* resulting from changes in environmental quality describes the socioeconomic impact of changes of the environmental state resulting from the pressure on the environment.
- The social *Response* to those changes in the environment refers to responses by groups (and individuals) in society, as well as government attempts to prevent, compensate, ameliorate or adapt to changes in the state of the environment.

Subsequently, the options' performances are calculated by applying simulation models and other elaboration procedures. The results are used to quantify the performance of every alternative option (i.e. alternative policy scenarios) in terms of the criteria selected. Analyses are carried out to evaluate the performance for each decision criterion. In the phase of evaluation of options, the *choice phase* has the aim to choose the appropriate measures by using multi-criteria analysis evaluation techniques.

An advantage of this method is the introduction of *environmental indicators*. Communication is the main function of the indicators. They enable or promote the information exchange regarding the issue they address. E.g. body temperature is an example of an indicator people use to measure health. Likewise, environmental indicators provide information about phenomena that are regarded as typical for and/or critical to environmental quality.

In this method, model selection is carried out through the use of the so-called D-P-S chains, which presents the causal links between *D*riving forces, *P*ressures, changes in *S*tates. However, it is not clearly addressed in relation to model selection. In addition, similar to other approaches, uncertainty and complexity do not receive sufficient attention in this approach.

2.2.1.4 Rapid-assessment modeling (De Kok and Wind, 2002)

Few studies have attempted to develop an appropriate modeling approach addressing the issues of model selection in the context of overall system consistency (De Kok and Wind, 2002). In order to design a model of a real water system that is appropriate for describing changes in the objective variables without being overly detailed or excessively coarse, the so-called *internal consistency* approach is proposed. The idea is that an integrated system is considered to be internally consistent if the level of detail of each process model is appropriate with respect to the accuracy required for interpreting the output of the system. Taking into consideration the interaction of each process model with the other models, each process model should be neither overly coarse nor exclusively detailed. To achieve such a goal, the design phase consists of two steps, namely a *qualitative* analysis and a *quantitative* analysis. In De Kok and Wind's approach, the qualitative analysis results in a conceptual framework of analysis, which links measures to objectives based on relevant processes, variables, and parameters. During the quantitative analysis, models and data are collected to quantify the system relationships.

The internal consistency concept provides a systematic approach for the design, and development of a DSS with appropriate selection methods. For relevant processes and variables, a design tree (Nieuwkamer, 1995) based on backward causal reasoning from the objective, similar to the general system analysis approach, is used. Aiming for reducing system complexity, a strength-weakness analysis is applied to distinguish between processes that have a significant influence on the management objectives and the uncertainty therein, and processes that do not. The strength-weakness analysis can both be based on qualitative and quantitative information. Models are formulated based on internal consistency concepts considering causal links between processes and uncertainty constraints.

To further the development of DSS, De Kok and Wind discussed some important aspects that should be taken into consideration for appropriate modeling. This was done by comparing six DSS design projects, related to different objectives in water resources management. The authors concluded that a successful DSS can be obtained when: i) a solid analysis of the problem from an integrated point of view precedes the design; ii) end users are actively involved from the beginning of the design; iii) a clear statement of purpose of the decision support system is available; iv) results are presented in a form tuned to the needs of the users; and finally, v) the design is flexible enough to cope with changing demand conditions. Drawbacks of practical model selection, which is often based on data and model availability, are pointed out as: i) it makes the design strongly case-dependent and difficult to become generic; ii) it causes confusion to the end users as the significance of the components is unidentified; more seriously, iii) some parts of the system can become too complex while other parts may suffer from a lack of detail, particularly when different modelers with different levels of knowledge contribute models and data. To avoid those problems, an *appropriate modeling* concept is proposed, that is to balance the model power (functionality of a model to distinguish alternative measures) and model complexity (Figure 2.4).

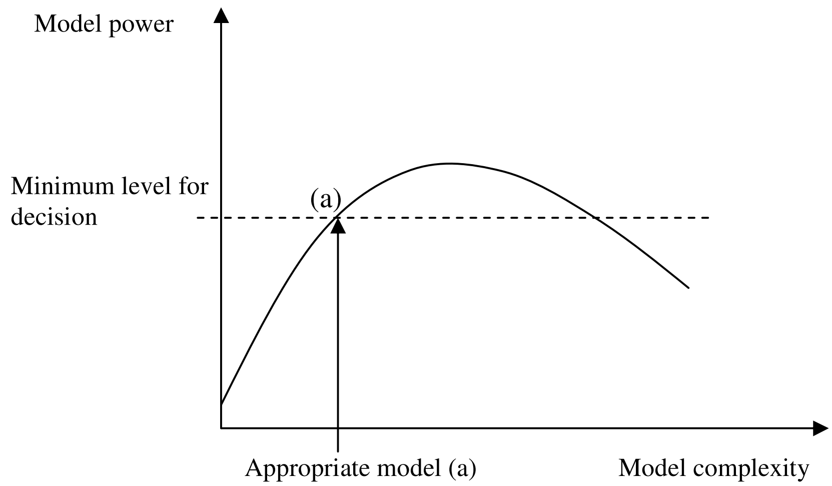

Figure 2.4 *Conceptual representation of appropriate modeling (De Kok and Wind, 2003)*

As shown in Figure 2.4, model complexity is assumed to follow a certain relationship with model power. Model power can be expressed in terms of accuracy or other criteria such as data demand or computing time, depending on what the end users are interested in. The requirement of model power has a minimum level that a model should achieve. In general, model power will increase with model complexity, thereby introducing more cause-effects relationships and variables. At a certain point, however, due to the high demands of data and computer skills such as numerical solutions involving more parameters, the extra uncertainty introduced by the higher model complexity may lead to a decrease in model power. Nevertheless, with a certain level of minimum model power, a certain level of model complexity may be met, which can be considered the appropriate model complexity.

The idea of appropriate modeling is indicative. It can be different when more considerations are introduced such as the modelers' experience or modeling skills. For example, model performance largely depends on the knowledge and experience the particular modeler has. A simple model may perform as well as a complex model, when used by modelers with rich experience and knowledge, while a complex model with sufficient data may perform relatively poorly if a less experienced modeler is involved. Meanwhile, the definition of a minimum level of model power, both qualitatively and quantitatively, is difficult. It can be a single criterion such as model accuracy or, more often, multiple criteria such as cost implications, data demand, interface flexibility, presentation methods and so on. Therefore, due to the practical limitations of model power definition, and different model performance by different modelers, such a method can only be indicative and difficult to apply in practice for model selection.

In De Kok and Wind’s work, a statistical method is proposed to select models in a more systematic way, taking an uncertainty perspective, which evaluates the model performance taking into account its uncertainty characteristics. The choice of a particular level of detail affects the model uncertainty, and hence its ability to distinguish between different management alternatives.

2.2.1.5 Open-design approach (Jamieson and Fedra, 1996a)

To develop a so-called fifth-generation hydroinformatics system (Abbott et al., 1991), Jamieson and Fedra (Jamieson and Fedra, 1996a, b; Fedra and Jamieson, 1996) presented the development of the DSS *Water-Ware* aiming for river basin planning using an *open design* approach consisting of three major steps, namely 1) the conceptual design, 2) components identification and integration, and 3) testing. A fifth-generation system aims not only at incorporating easy-to-use analytical capabilities, but also offers expert advice and intelligent integration facilities such as artificial intelligence and optimization techniques, so that end users do not need to have in-depth knowledge of all processes involved. The development procedures are presented in three papers, each describing one of the three development steps: 1) conceptual design (Jamieson and Fedra, 1996a), 2) determining planning capability (Fedra and Jamieson, 1996) and 3) formulating an activity involving example applications (Jamieson and Fedra, 1996b) (Figure 2.5).

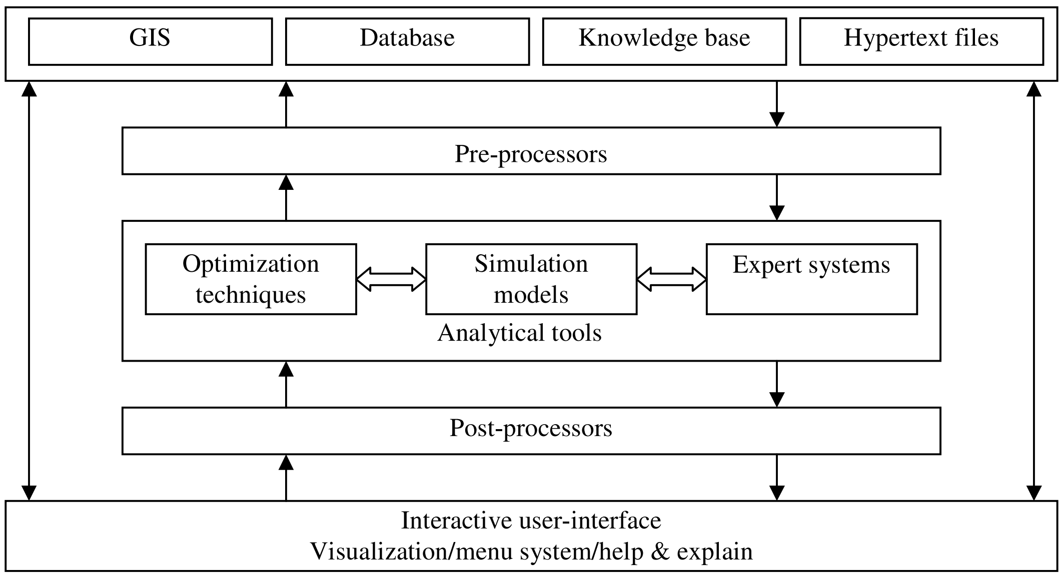

Figure 2.5 *System architecture for a fifth-generation DSS (Jamieson and Fedra, 1996b)*

In the conceptual design phase (*phase I*), the overall architecture of the DSS and the techniques involved, such as the use of GIS, and optimization techniques, is specified. In this design phase, the facilities of the DSS such as the river network editor, expert systems or library, or hypertext help function are identified based on user-friendliness. To deal with the problem in river basin planning, model components such as water demand forecasting, water resources planning, groundwater pollution control, surface water pollution control, or hydrological processes, are identified. To avoid difficulties of presentation method in relation to each of the module components, an independent river network presentation method is adopted. Care has been taken to enable an easy understanding of the system outputs by means of a user-friendly interface using hypertext to guide the end user and color graphics in presenting the results, interactive utilization of video player and other computer technologies.

Phase II determines the planning capability. The planning capability consists of the analytical solution to each component - both existing and intended - including: GIS, geo-referenced database, groundwater pollution control, surface-water pollution control, hydrological processes, demand forecasting and water-resources planning, as identified at the conceptual design phase.

Phase II describes the functionality and limitations of each model, as well as how they can be linked in the integrated system. The choice of models is based on a so-called rapid-prototype approach mainly based on evaluating the status of models ranged from completely ready to that needs to be developed.

Model integration takes place using the causal link relationship between each component. In the application stage (*phase III*) of the DSS, two river basins were selected to test the applicability of the DSS, namely the Thames basin in England and the Rio Lerma river in Mexico. Examples are given of real-world problems that can be addressed using this system, including water shortage, reservoir site selection, and decontamination of groundwater. An advantage of this approach is that models of various complexities can be developed and incorporated in the system.

Model selection in this approach is based on so-called *rapid prototyping* – which is a configuration of standard elements, taken from the available models and elsewhere – to provide a first impression of the proposed capability or layout which can be used as a template for producing more refined versions subsequently (Fedra and Jamieson, 1996). In other words, the models are collected without a critical analysis, based purely on their availability.

In contrast to a normal problem-based approach, the open-design method is based on the modelers' knowledge aiming at a generic DSS that can be used for any river basin planning. The development is largely based on experts' knowledge. End users' participation is taken into consideration in the design of a user-friendly interface, but the DSS is based on the assumptions of the modelers - assuming that all the design using graphical techniques are clear and understandable for the end users. Meanwhile, aiming for a generic tool, the design has taken a broad view of processes that can be included in any river basin planning. This can easily lead to overloaded complexity of the design which might be unnecessary for a specific river with specific physical conditions and problems. Model/data uncertainty and how the system design can benefit from UA are not discussed yet. In view of the nature of a DSS, uncertainty is always presented and this neglect limits the usefulness of the DSS. For example, future changes can be only assessed by considering uncertainty.

There is no emphasis on how to select an appropriate model; however, such a knowledge-based approach prevents missing of scientific principles. It might be overly complex due to the lack of specific requirements from the end users' point of view and the requirements from the natural conditions. For example, for a small river basin, the DSS may not cover all types of catchment ranging from very dry to very wet, associated with the existing climate characteristics. Therefore, it may not be necessary to include all kinds of hydrological models in the design of a catchment DSS, unless the DSS needs to be generic.

2.2.1.6 General characteristics of previous DSS design approaches

In general, a system analysis approach is followed for the design of a DSS for integrated RBM. This provides common steps of problem definition, management alternative identification, future context prediction, model and system formulation, and ranking and comparing alternatives for decision-making. Nevertheless, two different approaches can be discerned for the DSS design, known as the user-oriented and the knowledge-driven ones, which lead to different design architectures, i.e. to develop a DSS for specific problems and a particular river basin, or to develop a DSS as a generic tool that can be used in any river basin to deal with all kinds of problems.

The approach followed also determines the way of model selection. The user-oriented approach tends to make use of readily available models and data, whereas the knowledge-driven approach aims to develop models that are as complete as possible to cover any possible problems that may occur. Drawbacks are that a user-oriented approach puts the emphasis on the participation of end users, which may lead to an ill-structured design due to insufficient knowledge involved, whereas the knowledge-based approach assumes what might happen in the real world, which might lead to an overly complex system for a certain river basin. Nevertheless, both methods try to achieve appropriate modeling from a different perspective and in fact complement each other in the development of a DSS.

All DSS design approaches addressing the selection of models and variables should take into consideration the causal linkages between processes and variables to establish consistent model integration. However, what has not been clearly addressed so far is how models should be selected with minimum complexity but that still satisfy the functionality requirements of the DSS, i.e. the conceptual of appropriate modeling (De Kok and Wind, 2003). Moreover, the selection of models in the development of an integrated DSS is still not supported with a sound principle. This thesis does address this (see Section 2.4).

2.2.2 Issues addressed in the development of a DSS

In the development history of DSSs, various issues have been addressed to obtain a successful DSS according to requirements from end users and the purpose. These issues are policy analysis (Parker, 1995; Green and Kalivas, 2000; Silva et al., 2001; Verbeek and Wind, 2001; Green et al., 2002; Fassio et al., 2004), dealing with changes of and social opinion (Schielen and Gijsbers, 2003), and changes in information technology (Singhroy, 1995; Aerts et al., 2000; Halls, 2003; Morari et al., 2004). In the development procedure, the end users' participation is emphasized (Herrick and Jamieson, 1995; Newman et al., 1999; Farrell et al., 2001; Bell et al, 2001; Mostert, 2003; Nunneri and Hofmann, 2004; Mysiak et al., 2005), as it is important for a clear problem definition and benefits the development of a DSS. Difficulties involved in the development of DSSs, particularly in model selection and model/system performance evaluation, are also addressed. The following sections discuss the above issues sequentially.

2.2.2.1 Policy analysis

Policy analysis can be broadly defined as *the study of the nature, causes, and effects of alternative public policies* (Nagel and Neef, 1980). It has been widely applied to evaluate the effectiveness of policy in water resource management (e.g. Abrahamse et al., 1982; Parker, 1995; Giannias and Lekakis, 1997; Green et al., 2000). Defining appropriate policy or strategy can be difficult (Parker, 1995; Green et al., 2000), as a result of 1) the differences in nature among river basins concerning their geometry and climate properties, 2) the state of development of the country, and intensity of use of the floodplains. For that reason, the flood mitigation policy changes for Europe may not be applicable to rivers in Asian countries. For instance, in The Netherlands, the government's attitude towards floods has changed from 'fighting with floods' to 'living with floods', which has led to the policy of 'room for the rivers' (Silva et al., 2001). However, this policy cannot be applied in densely populated river systems such as the Yangtze River in China. In the Yangtze River basin, people are living along the river and are protected by the dikes. Here, flood management very much relies on the operation of flood defense systems such as dikes and/or retention basins, and making room for the river is simply impossible (e.g. Yin and Li, 2001). In summary, flood mitigation policies are region-dependent.

As Green et al. (2000) stated, there are no universal appropriate solutions. To find an appropriate policy, they recommended beginning by analyzing the nature of the flood problem in the area, then identifying the available options, and comparing these in terms of their contribution to the society's objectives (Green et al., 2000), or system analysis approach. Nevertheless, an appropriate integrated DSS should be able to deal with policy variations through spatial integration and should be able to lead to the implementation of a sustainable policy, i.e. integration of policies in the temporal domain.

2.2.2.2 Dealing with physical changes and social opinion variations

Schielen and Gijsbers (2003) pointed out the problem of clarifying end users' priorities and conditions during the design of a DSS. Thus, a successful DSS development approach should be able to deal with those changes. For example, in addition to the noticeable physical changes such as climate change, changes due to land use development, changes in the geomorphology, and policy change, which is reflected by the implementation of alternative measures, are beyond the control of present decision-making, and should be foreseen during conceptualization of the functionality of a DSS. Changes involved in the social opinion and knowledge towards IRBM also have a significant influence on the development and implementation of a DSS (Schielen and Gijsbers, 2003). One should be aware that although the design of a DSS is driven by the end users' requirements, the effectiveness of the DSS depends also on knowledge of how to use it. To solve such a problem, knowledge can be made available and understandable to end users by means of education and training processes during the development of a DSS. In addition, developments in computer science and GIS technology are essential for an interactive and user-friendly DSS and should be paid attention to, particularly in the conceptual design phase. The role of GIS has recently been drawing more attention and plays another vital role in visualizing scenarios for rapid assessment of large rivers (Singhroy, 1995; Aerts et al., 2000; Schielen and Gijsbers, 2003; Halls, 2003; Morari et al., 2004).

In short, an appropriate DSS should be able to deal with changes occurring in different aspects in relation to integrated RBM and the development of a DSS.

2.2.2.3 The role of participation by end users

A clear and correct problem and objective definition is essential for the development of a DSS. As Farrell and co-workers (2001) pointed out, '*one of the most important parts of this (*a DSS*) interface is that quantitative modeling efforts must be designed to answer the questions that decision-makers ask*'. Clearly, scientists or modelers need to respond to the issues of greatest value to decision-makers and stakeholders. As has been widely acknowledged, the involvement and participation of end users benefits RBM and the development of a DSS greatly (e.g. Mostert, 2003; Nunneri and Hofmann, 2004; Mysiak, et al., 2005). This requires an iterative communication process between end users and modelers, involving improvement of the awareness of knowledge of end users.

Benefiting from the progress of the end users' knowledge on DSSs, a DSS is a never-ending evolutionary procedure: problems can 'grow' and objectives can be changed during the development process. According to the experience of Matthies et al. (2003), in the development of a DSS for the Elbe River in Germany, technical requirements such as presentation methods, or impact assessment indicators or criteria, are self-evolutionary aspects of the design, which keep changing with the improvement of the end users' knowledge towards the end of the DSS project. This has also been the experience in other studies, for instance the one by Newman et al. (1999)

who reported that a DSS became obsolete because the end users (after having employed it for a short while) became familiar with its logic and were able to apply the DSS on their own. During the design of a DSS, the knowledge of the end users develops as they are educated through communication with modelers. Their understanding of the physical processes and models rapidly improves.

However, this is not always the case due to the occurrence of cross-links and feedbacks, and to difficulties in the communication between modelers and end users. The reason is that not many end users or stakeholders have a clear conception of what they would need in terms of technical requirements due to the distance between them and science and technology. As pointed out by Lankford et al. (2004), there is a high chance that insufficient scientific awareness on the part of a policy maker or decision-maker would cause a less effective implementation of a DSS, which supports the view that '*it is a mistake to assume good science can always provide a right answer for science-based policy dispute*' (Herrick and Jamieson, 1995).

2.2.2.4 Model selection

As stated in the famous Ockham Razor principle '*entities are not to be multiplied beyond necessity*'. This underlines an aspect of all scientific modeling and theory building that one should not increase, beyond what is necessary, the number of entities required to explain anything. This principle has been implemented in recent literature on model selection, in which parsimony or simplicity is balanced against goodness of fit (e.g. Grijspeerdt et al., 1995; Venterink and Wassen, 1997; Wood et al., 1998). This approach can be regarded as comparative, that is, the performance of alternative of models is compared to choose the most appropriate one meeting the best the defined criteria. Ad-hoc model selection based on the availability of models (e.g. Bathurst et al., 2003; Jeunesse et al., 2003) is not recommended in this thesis when models are not any more true/false, and are available at all kinds of institutional organizations and can be either commercially available or cost-free.

The comparative approach for model selection has been widely used. For example, Grijspeerdt et al. (1995) compared several one-dimensional sedimentation models. The models were evaluated with several *a posteriori* model selection criteria such as accuracy. The practical applicability of the models for the available data sets was also investigated. The final choice was then made for the one that provided the most reliable results. Venterink and Wassen (1997) compared six models predicting the vegetation response to hydrological habitat change. Wood et al. (1998) compared sixteen catchment models participating in a project for the comparison of land-surface schemes using ten years of forcing data for the Red-Arkansas River basins in the Southern Great Plains region of the United States. The comparison showed that there were variations for the performance between models of different structure. It was obvious that different models performed differently for a given system and that selection based on performance seems logical when there are alternatives of models to choose from.

To carry out a comparison, model evaluation is necessary. Mathematical models are in general characterized by a certain degree of uncertainty, resulting from the model uncertainty, such as the numerical errors (e.g. truncation error of numerical scheme) corresponding to model complexity and observation errors. Commonly used approaches of model evaluation are model uncertainty analysis and sensitivity analysis (Grieb et al., 1999; Mckay et al., 1999; Gustafsson and Mäkilä, 2001). UA and SA have been widely discussed in different disciplines, and are believed to be the most important tools to assist in the model selection process (Klepper, 1997; Hanna et al., 2001; Crosetto et al., 2001). The techniques of UA and SA are described by Morgan and Henrion (1990) and Saltelli et al. (2000).

It is obvious that UA and SA are important for model suitability assessment. However model uncertainties and model sensitivities are not the only two criteria for model selection. In general, a more complex model has a lower uncertainty and higher sensitivity, but it has a higher requirement with respect to the quality and quantity of the input data. Also, it is computationally intensive, which increases the model cost, and it might not be the most appropriate model.

This point has been elaborated by Snowling and Kramer (2001). Their work has primarily considered model performance as a whole for model selection. Assuming relationships between model complexity and other model aspects as shown in Figure 2.6, the model complexity spectrum assumes that data requirement, model flexibility and model sensitivity increase with the increase of model complexity, whereas the model error (or model uncertainty) responds in the opposite way.

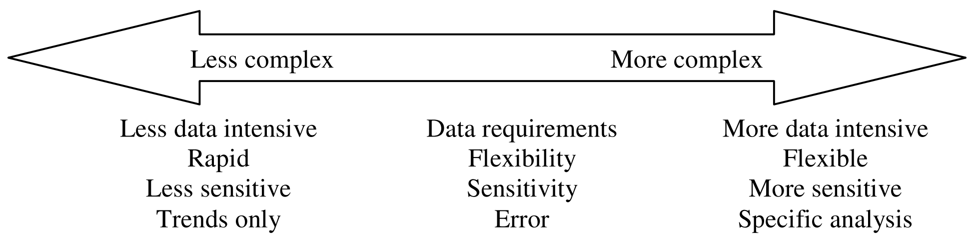

Figure 2.6 *Modified model complexity spectrums (Snowling and Kramer, 2001)*

To evaluate model performance, Snowling and Kramer (2001) proposed the use of model utility, which takes model sensitivity and uncertainty into consideration and expresses model usefulness quantitatively. By establishing the relationship between model utility and complexity, the most appropriate model is identified. Snowling and Kramer also defined the model utility as the function of individual model sensitivity and uncertainty, which quantifies model usefulness combing sensitivity and uncertainty. In addition, it can be used when model selection is needed to be made among alternative of candidates. The difficulty here is the quantification of the two weighting coefficients involved in the expression of model utility.

In addition, model selection is not equal to the selection of the modeling-software package. In general model, selection consists of two steps: identification of the conceptual requirement followed by the selection of a modeling-software package that can be used to set up the model. For each model with a certain complexity, there may be multiple software packages available, including commercial software or packages developed by institutes. Existing inventories of modeling software packages – such as the list of hydraulic models that can be used for flood hazard simulation (mapping) provided by the Federal Emergency Management Agency (FEMA) of USA (http://www.fema.gov/) (Appendix I) – can be really useful. Guided by such qualitative descriptions, the selecting of an appropriate modeling package can be made according to other criteria such as cost, user-friendliness, and supporting operation system. Such existing inventories, however, are not the ultimate solution as old software is often updated and new software packages are introduced and it's difficult to keep such an inventory up to date.

In summary, for model selection, the comparative approach might be effective when there are alternative models that function equally in terms of discipline but differ in terms of complexity and other aspects such as data requirement. However, when models need to be developed, such an approach is no longer applicable. Moreover, the comparative approach for model selection is a research activity rather than a practical solution for the development of a DSS. This makes it significant to search for a model selection method without unnecessary posterior quantitative analysis.

2.2.2.5 Uncertainty analysis

Uncertainty analysis has been found important to evaluate the overall performance of the models and the integrated systems (e.g. Snowling and Kramer, 2001), as well as to distinguish between the impacts of alternative measures (e.g. De Kok and Wind, 2003).

Various methods have been used to evaluate the usefulness of a DSS (e.g. Finlay and Wilson, 1997; Mahmoud and Garcia, 2000). However, those methods might be impractical due to various limitations. For example, an objective assessment is difficult to obtain due to subjective weighting or ranking method applied in multi-criteria analysis (e.g. Mahmoud and Garcia, 2000; Beynon, 2002). Another concept of system evaluation is to assess system performance using the concept of validation (Finlay and Wilson, 1997). In this method, the performance of a DSS is assessed by determining the validity of each component, based on aspects such as data, model accuracy, interface, robustness, and operational validity. Similar to multi-criteria analysis, this approach requires defining various criteria (named *validity* by Finlay and Wilson, 1997). The validity of most DSS aspects such as interface, robustness, and to some extent, the model accuracy, is subjectively defined and is arbitrary and case-dependent. Meanwhile, when a validation concept is applied for model selection, attention must be paid to extend the validation coverage. Nevertheless, due to the difficulties involved in these methods, UA shows its advantage for it is a well developed approach (e.g. Morgan and Henrion, 1990; Saltelli et al., 2000), and has been found effective in providing overall performance of models (e.g. Snowling and Kramer, 2001; De Kok and Wind, 2003).

Evaluation of model and system performance is important for model selection. For example, Snowling and Kramer (2001) introduced a term to present model usefulness as *utility index* which combines uncertainty and sensitivity and expresses the model performance as a unit. Despite the drawback of subjective assumptions involved in the determination of such an index, Snowling and Kramer concluded that a more complex model does not necessarily perform better. This conclusion is a very strong message which clarifies that it can be a mistake to assume that higher complexity always performs better.

Uncertainty analysis can also benefit the usefulness of a DSS by providing distinguishable results of alternative measures (De Kok and Wind, 2003). The distinguishability of the DSS can be presented by the comparison of uncertainty distributions of each comparable scenario. This can be a useful criterion to judge whether the system performance is appropriate. That is, if the difference is clearly distinguishable, the model or system structure might be of sufficient sensitivity (De Kok and Wind 2003).

Thus, an appropriate design of a DSS should include UA to not only estimate the general performance of the DSS, but also provide distinguishable and comprehensive presentation of the outcomes.

2.3 State of the Art of Flood Risk Assessment

To support flood and risk management, flood risk assessment aims to predict the effectiveness of flood management measures in terms of environmental impacts and/or socio-economic effects.

The concepts of risk assessment and management provide the basis for decision-making with regard to individual risk management measures, and also with regard to a whole, *integrated* programme of measures. They enable the following key questions to be addressed when determining policy, strategic planning, design for construction decisions (MAFF, 2000):

- What might happen in the future?
- What are the possible consequences and impacts?
- How possible or likely are different consequences and impacts?
- How can the risks be managed?

This has much to do with the understanding of the term of *risk*. Risk has a range of meanings and multiple dimensions relating to safety, economic, environmental and social issues. These different dimensions often reflect the needs and risk perception of particular decision-makers and as a result there is no unique specific definition of risk and any attempt to develop one would inevitably satisfy only a proportion of risk managers.

It is common, however, to describe risk as a combination of the chance of a particular event with the impact that the event would cause if it occurred (NRC, 2000; Vrouwenvelder et al., 2003), which forms the current FRA methodology, with variations according to which indicators are required by risk managers.

2.3.1 Flood management approaches

Flood disasters are identified as the world's most frequent and damaging types of disaster and their occurrence appears to be increasing (Glickman et al., 1992; Parker, 1996). To cope with or to prevent floods, flood management is needed.

Flood management approaches have changed throughout history and four generations of flood management programmes can be discerned (Green et al., 2000). The *first* generation pertains to the small-scale, local adaptations to make people more resilient to flood hazards and disasters, for example, building houses above anticipated flood levels, and using floating houses. History suggests that as modernization takes places (i.e. through urbanization and economic growth) indigenous approaches such as embankments are eroded (Parker, 1996).

The *second* generation approach, characteristic for the late 19^{th} and most of the 20^{th} century, was the 'engineering' approach (Green et al., 2000), such as building a dike along the river. It is also called the *structural approach*. The philosophy was strongly rational: rivers were being 'trained' or 'improved' to become efficient and to stop floods from interfering with human activities. However, structural approaches have a number of disadvantages:

- Flood control structures may encourage further floodplain development.
- Flood embankments may only be partially effective in exceptional floods (i.e. they may be overtopped or breached).
- There may be negative impacts on downstream areas (making their flood problem worse).
- Flood control may only address a part of the problems which cause flood disasters without addressing people's vulnerability to flood hazards, not to mention that improvement of flood defense works is an endless process, due to the climate changes and socio-economic changes within the river basin.

The *third* phase was the advocacy of *non-structural approaches* (Penning-Rowsell and Fordham, 1994; Vis et al., 2003; Dijkman, et al., 2003; Nishat, 2003). Instead of engineering the rivers, the approach centered upon people's awareness. For example, planning control may be proposed to prevent extending the built-up area to the floodplains, as well as small-scale structural modifications of individual buildings and measures to move people away from areas at risk. Non-structural approaches were generally assumed to offer an alternative to, and to be a replacement for, traditional engineering approaches. Although some past investments in flood control structures proved to be wise, many structural and non-structural strategies have failed to be sufficiently effective, and that the non-structural model of flood management, so strongly advocated in the United States, requires rethinking (Changnon, 1996; Mileti, 1999; Myers and Passerini, 2000). Rethinking is required because of its inadequacies in the United States and its poor applicability to many other world regions.

Driven by the concept of sustainable development as well as integrated management of a river basin, the holistic approach, or the integrated approach, regarded as the *forth generation*, may be seen as the solution which brings together all types of approach aiming for sustainable flood management (e.g. Gardiner, 1994; Blaikie et al., 1994). This approach can be referred to in terms of 'flood alleviation' and 'flood risk mitigation', rather than in terms of flood control or flood risk management. The implementation of such an approach implies developing a mechanism for extending low-cost loans to inhabitations of low economic status and may be the kind of strategy which holds the best promise for the future in terms of making people more resilient to the effects of floods. This approach leads to an emphasis on sustainable and integrated catchment management, on the wise use of floodplain and coastal zones, on empowering local communities to make choices about land development and flood alleviation, on reducing the impact of humans on the environment, promoting flood disaster resilience, on valuing and preserving the best of indigenous adaptations, and improving public awareness of risk and capacity to respond. Consequently, the implementation of measures to achieve the objectives listed above, integrated FRA (IFRA) becomes essential.

IFRA can be defined as the FRA aiming for integrated flood management involving both long-term planning and short-term operations, on both local and catchment scales. IFRA is needed to assess the validity for transferring management approaches, or policies, from one region to another, due to spatial change, as explained in section 2.3.1. An effective implementation of policy or management measures requires an IFRA to assess its effectiveness.

2.3.2 Measures of flood management

Typically, there are two ways to deal with flood risk: by means of structural measures (e.g. White and Haas, 1975; Thampapillai and Musgrave, 1985) and by non-structural measures (e.g. Changnon, 1985; Edward and Thomas, 1993). Structural measures are engineering works for the protection of reservoirs, dikes, flood protection walls, retention basins and so on, and for channel improvement. The non-structural measures usually reduce the flood damage, and can be applied to all hydrological zones. These organizational, financial and regulatory policy measures include land use planning and zoning, flood prediction and warning systems, evacuation and rescue programmes, and flood insurance or compensation programmes.

Temporally, flood alleviation measures can be categorized into *pre-flood* (or preventive) measures, emergency (or operational) flood management, and *post-flood* (corrective) measures (Pols, 1995; Kundzewicz and Samuels, 1999). Table 2.1 listed the categorization of measures in these two dimensions.

Table 2.1 *Flood alleviation measures versus flooding stage (Pols, 1995)*

Measures	Preventive measure (before flood)	Emergency measures (during flood)	Corrective measures (after flood)
Structural measures	Dikes, storm surge barriers	Sand-bagging, emergency repair of dikes	Repair, restoration and reconstruction
Non-structural measures	Land use zoning, flood proofing	Warning system, evacuation, rescue	Insurance, relief and rehabilitation

The main activity in flood management is dealing with flood risk. As described previously, approaches of dealing with flood risk vary through history. From simple adaptations such as house construction on a higher level, to engineering approaches such as the construction of a dike, and recently the introduction of non-structural measures such as the use of retention basins, as well as the emergence of the integrated flood risk management approach to mitigate flood risk to the minimum (Green et al., 2000). It is necessary to re-assess the adopted management approach constantly and to transfer knowledge from one country or area to another. However, very often rather than knowledge transfer taking place, the approach developed in the context of one country or river has simply been transposed to another country or river, which might be inappropriate due to the differences in the nature of the two cases. The knowledge should be transferred, rather than the specific measure. The insight into the usefulness of knowledge transfer requires an integrated FRA.

2.3.3 Flood risk assessment approaches

Flood risk assessment is an approach that assesses risk and damage caused by flooding. Since its introduction (White, 1945), FRA has taken many forms through history. Two major approaches can be discerned in FRA: a statistical approach, and a deterministic approach or physically-based damage assessment. These two types of FRA have been developed and applied worldwide (e.g. CUR, 1990; Stedinger, 1997; Todini, 1999; Vrijling, 2001; Horritt and Bates, 2002; Van der Sande et al., 2003; Sinnakaudan et al., 2003).

Following the classical mathematical risk concept (Vose, 1996), the *statistical* approach aims to predict the expected value of the annual flood damage (e.g. Arnell, 1989; Stedinger, 1997). In this approach, risk is defined as a measure of the danger that undesired events represent to human beings, environmental and economic values. It is expressed in terms of the probability, such as distribution of annual flood peaks, and the consequences of the relevant undesired scenarios, such as the inundation depth, environmental impact, and socio-economic loss. Mathematically, risk is calculated as the product of probability and consequence, expressed as:

$$R = P \times C \tag{2.1}$$

where R denotes risk, P denotes the probability of failure such as occurrence of overtopping of a dike, and C denotes the consequence, e.g. the damage and loss of life.

This approach has been used frequently in flood risk management. In The Netherlands, the well-known Delta Works flood defense system was developed after a serious coastal flood in 1953 (CUR, 1990; Vrijling et al., 1998; Rhine Atlas, 2001). Based on a probabilistic design, the Delta Works resulted in the construction of the most important dike ring to protect Holland, with a design return period of 10,000 years (Parmet, 2003). In the USA, a study has been carried out to examine flood risk management for American rivers based on the annual expected damage (Stedinger, 1997; NRC, 2000). In Italy, the development of an operational decision support system for flood risk mapping, forecasting and management used a model based on the expected damage concept (Todini, 1999). In England, flood risk has been studied intensively since 1970, illustrated by the initiation of a Flood Hazard Research Centre (FHRC) at Middlesex University, which has resulted in thirty years of flood hazard studies including both the statistical and the physically-based approach (e.g. Penning-Rowsell and Parker, 1987; Fordham et al., 1991). In general, FRA is based on the statistical concept.

With the development of computer-based simulation and GIS tools, the consequence of the risk-analysis based approach, namely the calculation of the water depth and water velocity in the affected area, becomes more and more sophisticated and comprehensive. Referred to as *physically-based* damage assessment, this approach in general requires a complex inundation computation using 2D hydrodynamic models to determine the maximum inundation depth (e.g. Van der Sande et al., 2003; Horritt and Bates, 2002). The aim is to calculate the direct economic loss due to inundation caused by a specific flood event, for example, the damage caused by a 100-year flood. The physically-based FRA is more recent and received increasing attention during the past ten years. With the emergence of remote sensing technology, and the progress in computer technology in the 1990s, substantial computations to support physically-based flood damage assessment using complex inundation models became feasible (e.g. Van der Sande, 2001; Sinnakaudan et al., 2003). Based on comprehensive hydraulic modeling, the physically-based FRA is able to assess risk alternatives through the analysis of flood mitigation measures, such as the use of upstream retention, or an intentional dike break in an economically less valuable area. Some physically-based FRA studies confused risk (or hazard, or damage) maps with inundation maps (e.g. Shidawara, 1999). For clarity we distinguish between inundation maps – the direct results of inundation modeling – and damage/risk maps – the final outcome of a physically-based FRA.

Whereas the statistical approach has been widely studied and applied in flood management throughout the history of FRA for long-term planning, the physically-based FRA is used preferably for short-term analysis and implementation of flood mitigation measures.

2.3.4 Flood damage functions

Flood damage functions form the core of FRA. However, due to the complexity and uncertainty involved in the estimation of flood losses, it is difficult to obtain a generic and accurate flood damage function, for the following reasons. First of all, flood loss estimation requires substantial resources, which are rarely obtained. Secondly, loss is usually reported in different ways, and the distinction between direct and indirect damage complicates the problem (Burby, 2001). Thirdly, the area-dependent economic situation makes it difficult to directly transfer flood damage functions to other areas (Van der Sande, 2001).

Although it is difficult, there are ways of establishing relationships between flood damage and effect factors such as inundation depth and flooding frequency. A commonly used method is the flooding frequency ~ damage relationship curve. The derivation of an empirical frequency ~ damage curve can be found in the literature (Oliveri and Santoro, 2000; Shaw, 1994). It can be established under the simplifying hypothesis that the damage return period is the same for the event from which the damage arises, and can be estimated as a function of water depth, given socio-economic conditions. The most widely used method is the depth ~ damage curve where the damage is expressed as a function of the inundation depth associated with land use types or property categories (CUR, 1990; Kok, 2001; Vrisou van Eck and Kok, 2001).

However, it has been found that large differences exist in the depth ~ damage relationships as a result of different methods of categorizing land use and estimating damage loss, as well as differences in the economic situation of regions. In order to obtain comprehensive flood damage functions, various aspects should be taken into consideration. The most important steps are (i) categorization of land use and the damage associated with inundation depth, (ii) determination of the maximum damage expressed in monetary terms, which is largely depending on the location of the flooding area, and (iii) including important economic parameters such as inflation rate and exchange rate. Uncertainty of flood damage functions is therefore mainly related to the different sources of flood damage functions. In summary, collecting adequate flood damage functions is essential for a realistic FRA expressing with economic loss.

2.3.5 Velocity effect

It is known that flood damage is not only affected by the inundation depth, but by many other parameters such as the inundation duration, flow velocity, wind direction and wind force, apart from non-hydraulic factors like pollutant and sediment transportation, or anticipatory behavior of the people and companies in risk areas. However, an important factor as the flow velocity, which may cause additional damage (for example in steep areas), is rarely included in FRA studies. If the river is very deep and flows fast, it could wash away buildings and property and also drown people. At successively lower discharges corresponding to lower water velocities and shallower depths, the water becomes relatively safer. At low water depths, fast-flowing water may be less life-endangering, but could still result in damage to property. Nevertheless, the absence of flow velocity can prevent FRA from being appropriate.

A recent flood in Cornwall, England showed the significance of damage caused by flow velocity. On 16 August 2004, a 75-mm rainstorm of two hours caused two rivers flowing through the village of Boscastle in south-west England to flood their banks, and the rapid flow swept away buildings and cars (BBC Cornwall, 2004). Clearly, flow velocity should be taken into consideration for a proper FRA in these situations. This is also true for other parameters, such as the inundation duration which can be significant for agricultural production. However, it has been found difficult to quantify such important parameters.

Several studies have been carried out to quantify the relations between velocity and damage (Stephenson, 2002; Roos, 2003; Asselman and Jonkman, 2003; Kelman and Spence, 2004). To obtain a quantitative expression between velocity and flood damage, Roos (2003) studied the collapse of buildings caused by velocity. It was found that a velocity higher than 2.0 m/s together with a depth more than 0.5 meter will result in damage to buildings and that also applies to a water depth that exceeds 1 m with a velocity of more than 0.1 m/s. This demonstrates that damage to buildings is caused by a combination of inundation depth and velocity. There are combinations of water depth and flow velocity for which a wall of a certain building type will collapse with 100% probability. Based on this relationship of water depth and velocity, Asselman and Jonkman (2003) developed a damage model to simulate the loss of life assuming the collapse of a building in rapidly flowing water will result in the death of all those present in the building. Such an expression is limited to buildings only, and cannot be applied to other land use types such as crops and roads.

Kelman and Spence (2004) recently proposed a different quantitative expression for the effect of velocity on flood damage. The method is based on knowledge of the underlying hydrostatic and hydrodynamic processes to derive a value of economic loss. However, this method remains purely illustrative and qualitative, and needs to be validated before it can be used. In summary, quantification of damage ~ velocity remains limited.

In view of the low number of relationships that have been investigated (only the effect of flow velocity on buildings), it is meaningful to seek a generic method which can take velocity into consideration. Among various flood damage evaluation methods, the concept of risk level index is found to be interesting (Stephenson, 2002; Fattorelli et al., 2003; Vrouwenvelder et al., 2003). This method combines depth-caused damage with velocity effects into an index of risk levels using a predefined classification of velocity and inundation depth.

Stephenson (2002) adapted the flood hazard diagram to express the hazard associated with flow velocity. The diagram suggests that shallow depths can be countered by protective measures which could be implemented if there was a warning of an impending flood. In other less dangerous situations, very shallow flooding on roads could be minimized by vehicles avoiding those roads or traveling very slowly. This type of diagram has been adopted by various flood management organizations in for instance Minnesota in the United States of America (1969) and New South Wales in Australia (1986). The ranking of the hazard was simplified by Stephenson and Furumele (2001) in studies for eastern Gauteng (South Africa). However, the classification of risks with an index indicated as 0-3 lacks objective quantification of the effect of velocity on properties.

Similar studies were made on rural rivers (Du Plessis, 2000; Adriaans, 2001; Fattorelli et al., 2003). For example, in the risk assessment for risk reduction for the river Adige in Italy (Fattorelli et al., 2003) a criterion to identify four different levels of hydraulic risk (R4 to R1, with R4 identifying high risk and R1 low risk) was defined in agreement with the Adige River Authority, overlapping different classes of flood hazard with different classes of potential damage. Here, the flood hazard was defined from the combination of water level and velocity corresponding to return periods, whereas the potential damage is estimated as the economic value in monetary terms. This method introduced velocity into the assessment of flood damage. However, the potential damage is qualified in terms of four classes associated with four different combinations of land use classes, which does not give quantitative economic values for the properties.

The essential part of adapting the risk matrix method is the ranking method. Scales for probability and consequence should be similar and designed in such a way that the combination of probability and consequence reflect the desired weighting. A high- probability, low-consequence risk has the same significance as a low-probability, high-consequence risk. This can be the basic rule for the ranking of the variables. There are, however, no firm rules for combining probability and consequence. The Environment Agency's Guidance for Engineering Project Risk Management (Environment Agency, 1997), for example, reflects a risk-averse tendency where attention is focused on *High/High* risks, followed by all risks with *High* or *Medium* consequences. Any subsequent action will be based on the assessed benefits of risk avoidance compared with the cost of mitigation.

The important part of determining the consequence of combining probability distributions is the definition of the risk level. As discussed by Vrijling (2001), the risk index can be defined as the levels corresponding to actions or reactions where flood management activities such as evacuation take place. A similar idea was presented by Vrouwenvelder et al. (2003) who derived the risk matrix from a matrix normally used by commercial companies. In their study, the consequence class (damage in monetary terms) and flood frequency are combined. Based on that, four risk indexes are classified, associated with four actions as: no further action, optional action,

action at next occasion, and direct action. Such a risk matrix is very useful for short-term flood mitigation management such as evacuation.

In summary, due to the lack of quantitative expressions between velocity and damage/risk, the risk matrix can be a method to include additional damage caused by velocity. However, such a method may suffer the difficulties of determining risk indices and classification of damage indicators such as damage, inundation depth, and the combination with flow velocity.

2.3.6 Selection of appropriate hydraulic models

Hydraulic models represent the physical changes that are the major input for IFRA. Various aspects should be considered: complexity, uncertainty and sensitivity, flexibility, user-friendliness. In principle, a multi-criteria analysis can be used to select a hydraulic model independently. However, this requires assessing a score for each criterion, which will be subjective and does not guarantee the selection of the most appropriate model. The question arises whether there exists an alternative approach for the selection of hydraulic models for IFRA.

As discussed previously (section 2.3.4), common methods for model selection (e.g. Grijspeerdt et al., 1995; Venterink and Wassen, 1997; Wood et al. 1998) – including the selection of hydraulic models for FRA (e.g. Horritt and Bates, 2002) – are based on the comparison of performances among candidates. This method is not practical because in reality, no complete comparisons can be made among all possible modeling software packages. An alternative is to select modeling software packages based on the complexity identified by using causal relationships between processes and variables (EEA, 1999; De Kok and Wind, 2002; Fassio et al., 2004; Mysiak et al., 2005). Nevertheless, except the direct comparative approach give by Horritt and Bates (2002), none of these approaches address the selection of hydraulic models in relation to IFRA.

The key issue in developing the IFRA is the selection of hydraulic models. As described above, the two types of risk models have different requirements with regard to the hydraulic factors. The statistical approach requires water level ~ discharge relationships only as hydraulic computation, while the physically-based approach requires a hydrodynamic computation to obtain an inundation map. The obvious choice would be a full 2D model to satisfy both risk assessment approaches. However, due to the large data demands and computational loads, the complexity of setting up the model as well as in the calibration and validation uncertainties caused by the elevation data and roughness assumption in the floodplain, it is more time-consuming to use a 2D model to generate water level ~ discharge relationships. Moreover, the uncertainty is not less, particularly in the floodplain area where calibration and validation are difficult. In addition, due to the large computational loads of a 2D model, it might be difficult to apply the 2D model directly to provide a rapid risk assessment with short calculation time, which is essential to support decision-making for short-term flood management. Interpolation of a set of pre-calculated scenarios might help; however, the uncertainty of interpolation may be large. On the other hand, a simple steady-state flow model could be, conceptually, sufficient to provide a water level ~ discharge relationship. From a practical point of view, i.e. for a rapid assessment, it is possible to choose the simple steady-state flow model for the statistical risk-analysis based approach.

Thus, guided by the risk concepts, two types of hydraulic models are selected and applied in the establishment of the IFRA framework, namely a 1D steady-state hydraulic model that can provide water level ~ discharge relationships, and a 2D floodplain model that can simulate flow dynamics.

However, there is no unique standard that can be applied to judge the appropriateness of a modeling software package. As described in section 2.3.4, the selection of model concepts is just the first step of model selection. The next step is to choose an appropriate modeling software package. For inundation modeling, many packages are available, such as: SOBEK1D2D from WL|Delft Hydraulics (http://www.sobek.nl/), MIKE FLOOD from DHI (http://www.dhisoftware.com/), WAQUA from The National Institute for Coastal and Marine Management (RIKZ) (http://www.netcoast.nl/), FLOODWORKS from Wallingford (http://www.wallingfordsoftware.com/). These modeling software packages differ in structure, including numerical solutions and operation environment, which leads to different requirements for their implementation and integration with other model components in an IDSS. In addition, the experience of a modeler also plays a vital role in the performance of the model. Therefore, to select a modeling software package with known complexity in terms of mathematical concepts, the questions to be answered are: What are the available packages that satisfy the complexity requirements? How much does the model cost in monetary terms? Who is the provider? What limitations or advantages does the model have? What technical support can the modeler obtain? A qualitative description of available models can be the firsthand information for the selection of a particular modeling software package.

Ideally, the selection of a modeling software package should be based on an in-depth understanding of all possible available modeling software packages, but this is impractical in reality. The question is, *is it necessary to carry out a quantitative comparison for such an evaluation for all possible available models*? This can be true when the conceptual characteristics, i.e. complexity, of hydraulic models have been determined in advance qualitatively. This can be achieved through a controlled causal relationship searching. Here, controlled indicates multiple constraints involved in the selection of models such as input output requirements, data availability, and functionality requirements. Therefore, searching for an alternative method for model selection that avoids posterior quantitative comparisons is addressed in this thesis (see Section 2.4).

2.3.7 Uncertainty propagation

Uncertainty analysis is important for FRA. FRA is often used to support decision-making for flood and risk management (e.g. Pivot et al., 2002; Plate, 2002; Zerger et al., 2002; Chauhan and Bowles, 2003; Van Manen and Brinkhuis, 2004). The importance of UA can be seen at two phases. First, in the implementation of FRA, unawareness of uncertainty means it is risky to make a decision based on the FRA results. For example, although the differences between different measures are distinguishable, the results might be wrong due to the high uncertainty involved in either the model or the data. Secondly, in the development stage, UA is important to identify appropriateness of models including model complexity, sensitivity of variables and parameters, as well as uncertainty contributions.

To study uncertainty in FRA, two major steps should be involved: 1) identification and quantification of uncertainty sources; 2) propagation of uncertainties from data sources such as time series, hydrological statistics, DEM, land use data, and models including hydraulic models and flood risk models. Considering the multiple dimensions of flood risk including environmental aspects and socio-economic aspects, range of UA in relation to FRA is large, depending on the indicator that the risk refers to. Some works have been carried out to study some of the aspects indicated above (e.g. De Blois and Wind, 1996; Zerger et al., 2002; Chauhan and Bowles, 2003; Apel et al., 2004; Merz and Thieken, 2005).

In their study of identification uncertainty sources in flood damage assessment, De Blois and Wind (1996) have identified the most important uncertainty sources as (in sequence of significance): river dike height (for rivers with dikes), river discharge including its frequency of occurrence, the damage estimates, and the risk of dike breach. These conclusions are also confirmed by the results of the Monte Carlo simulation and experts' opinion. However, in this study uncertainty in hydraulic model structure was not taken into account, which limited the conclusions of the uncertainty contribution to flood damage. Zerger et al. (2002) compared uncertainty modeling techniques for storm surge risk management. The results show a significant effect of spatial uncertainty on flood risk mapping. Based on the UA of GIS raster data, i.e. digital terrain models, their work pointed out the importance of analyzing model uncertainty for decision-making in flood risk management. Chauhan and Bowles (2003) presented a framework for UA in dam safety risk assessment, including an approach to incorporating input uncertainty into the risk analysis model. Their work shows, in general, a significant benefit of UA for decision-making related to dam management, with additional information for the presentation of outcomes of risk, for example, by including estimates of the level of confidence. The study, however, focuses on statistical calculation, i.e. failure of a dam, excluding other variables such as geographical (DEM) and land use condition, which is important in risk assessment in RBM. In the quantitative FRA for polders, Van Manen and Brinkhuis (2004) assessed risk for a polder (low-lying area that used to be permanently under water, but was reclaimed). They have indicated that it is difficult to apply risk assessment tools due to the large uncertainties involved in the calculations. In their studies, uncertainties are found significant in the probability calculation, and the relation of flood scenario – damage/victims. For example, a sensitivity analysis showed that just a small difference in the relation of the water rise rate and victims resulted in 10 times more victims. An effective risk assessment therefore requires a comprehensive understanding of uncertainties, and an accurate flood damage function. In a flood frequency analysis, Merz and Thieken (2005) categorized uncertainty sources as two basic kinds: natural uncertainty stems from variability of the underlying stochastic process, such as annual rainfall, and epistemic uncertainty from incomplete knowledge about the process under study, such as model structure. They concluded that these two kinds of uncertainties should be separated. They also found out that epistemic uncertainty can be reduced by more knowledge, whereas natural uncertainty is not reducible.

These studies, however, merely looked at uncertainties related to certain dimensions of FRA. A *whole* view, which should involve different temporal and spatial scales, is not reflected.

Recently, Apel et al. (2004) proposed a framework for a comprehensive FRA chain involving processes by deterministic, spatially distributed models at different scales. To circumvent high CPU-time demands, they developed a stochastic flood risk model consisting of simplified model components associated with the components of the process chain. UA has been carried out for this simplified risk assessment model. The results show a significant uncertainty contribution from extreme value statistics, while a stage ~ discharge relationship proved relatively less important, and the damage module (the last module of the modeling system), does not influence the probability of flooding, and only alter the maximum damage caused by flood. The breach module was found to contribute a large uncertainty. The study provides an indication of uncertainty propagation through the chain of risk assessment. However, it is limited by the simplification of the modules. For example, the simplification of hydraulic modeling without the necessary 2-dimensional computation (section 2.4.6) may suggest a different order of the uncertainty contribution.

Nevertheless, as Vrijling (2001) pointed out, uncertainty is not only a property of the physical reality but also of the representation and extrapolation of human knowledge. As a consequence, risk assessment can be improved by increasing human knowledge. Apart from the data uncertainty, many other uncertainty sources are found to be significant for FRA. The most significant ones are the difficulties in quantifying uncertainty of variables such as the damage function parameters, and the uncertainty of model structure in terms of complexity. Meanwhile, to how complex the models should be built is also essential. As a result of the integration of hydraulic models, UA of the integrated system is more complex than the analysis of empirical quantities involved in each single model.

2.4 Appropriate Modeling for the Design of a DSS

A systematic approach is important in the selection of appropriate models, and in the quantitative evaluation of the system performance. Following a classical system analysis approach (Forrester, 1962; Miser and Quade, 1985), a design framework for DSS (Figure 2.7) is proposed based on the practical needs identified during the development of a DSS for the Elbe River in Germany (Matthies et al., 2003). The framework accounts for the internal consistency approach (De Kok and Wind, 2002), and uses environmental indicators (EEA, 1999) for model selection. With reference to the term appropriate modeling, the framework addresses in particular the issue of model selection and model formulation.

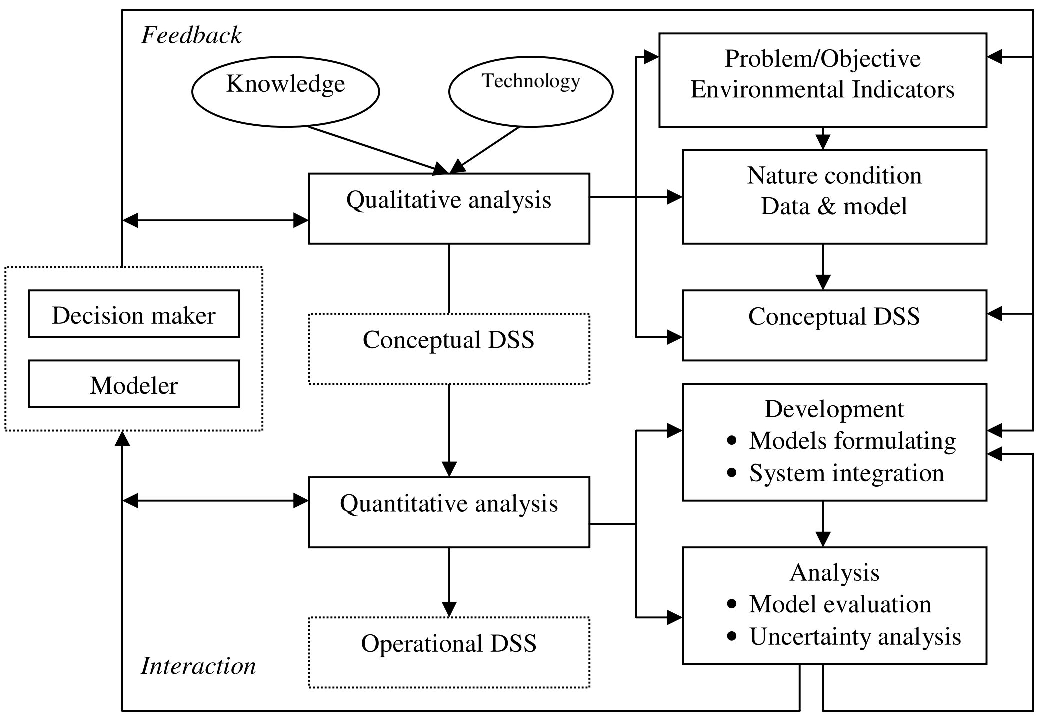

Figure 2.7 *Conceptual framework of analysis for appropriate modeling in the context of DSS development*

As shown in Figure 2.7, the framework consists of an iterative and interactive process to develop a DSS, pertaining both to the qualitative and quantitative design phase, involving feedback from end users (for defining clear functionality), and feed-forward analysis from modelers (facilitating the DSS with sufficient knowledge and technology).

Based on the work by De Kok and Wind (2002), two major phases are involved in developing the architecture of appropriate modeling, namely a *qualitative analysis* and a *quantitative analysis* phase. Qualitative analysis defines the problem and objectives, the environmental and/or socio-economic indicators, and the relevant processes that can be represented by models, and formulates a conceptual DSS that accounts for natural conditions in a future context. Quantitative analysis aims at developing the actual models that follow from the qualitative analysis. In addition, UA is applied to evaluate the performance of component models and the overall system. Both phases require intensive communication between modelers and decision-makers (or end users), which makes appropriate modeling an interactive and iterative procedure. During the development, scientific principles are firmly followed to assess and determine the complexity of the models, while new developments such as within GIS technology are applied extensively.

The following sections describe the various steps involved in the appropriate modeling, including the outcomes of each step.

2.4.1 Qualitative analysis

Aiming for a conceptual DSS, three steps are involved in the qualitative phase:

1. Formulation of problems and objectives, as well as environmental and socio-economic indicators;
2. Identification of relevant physical and socio-economic process and variables; and,
3. Formulation of the conceptual DSS framework that contains the identified models and variables.

Provided sufficient knowledge is present with the modelers, the qualitative phase essentially identifies all system components through a causal reasoning procedure. It is also important to identify the appropriate complexity without actually carrying out a quantitative analysis. The following sections elaborate the processes involved in the qualitative phase in some detail.

2.4.1.1 Defining problems

At this stage, the problem is first defined from the end users' point of view. Problem definition includes an outline of the management issues, potential alternative scenarios, legal and operational constraints, appropriate indicators and performance measures, and the information and functionality desired from the DSS that will be useful for decision-making.

2.4.1.2 Identifying relevant processes and variables

After the problem formulation, the measures and indicators are to be linked qualitatively, by selecting the relevant processes and variables, and taking the indicators as the starting point. Environmental and socio-economic indicators defined during the problem definition step can be used to guide the selection of processes and variables.

Before the identification of variables and processes, the modelers should be aware of the information flow through a river basin system, in terms of water flow directions, meteorology (rainfall or precipitation) → hydrology (rainfall runoff at catchment scale) → hydraulics (flow routing at the scale of the catchment or river reach including the floodplain area).

Several methods are possible to determine which processes should be included in a systems network (De Kok and Wind, 2002). Empirical reasoning is the most common approach. However, the drawback of this approach is that the disciplinary background and preferences of the modelers limit the scope of the system. Methods based on graph theory (e.g. Warfield, 1976; Bakker, 1987; De Vries, 1989) can facilitate the translation of scientific knowledge into system diagrams, yielding a system description in which the objectives and measures are explicitly present, however can result in a complex system network. With a backward causal reasoning approach, design trees (Wood et al., 1989; Nieuwkamer, 1995) can be used to construct an overview of all tentative alternatives within a hierarchical architecture. Likewise the graph theory, the tree can easily grow too large when there are too many processes involved. Another drawback is that the design tree can only represent available knowledge and does not help to acquire new/missing knowledge.

To cope with the drawbacks of the common methodology mentioned above, a so-called *double-direction searching* method is proposed (Figure 2.8). This method combines the backward causal reasoning based on environmental indicators, with a forward search based on the cause-effect relationships between data and processes.

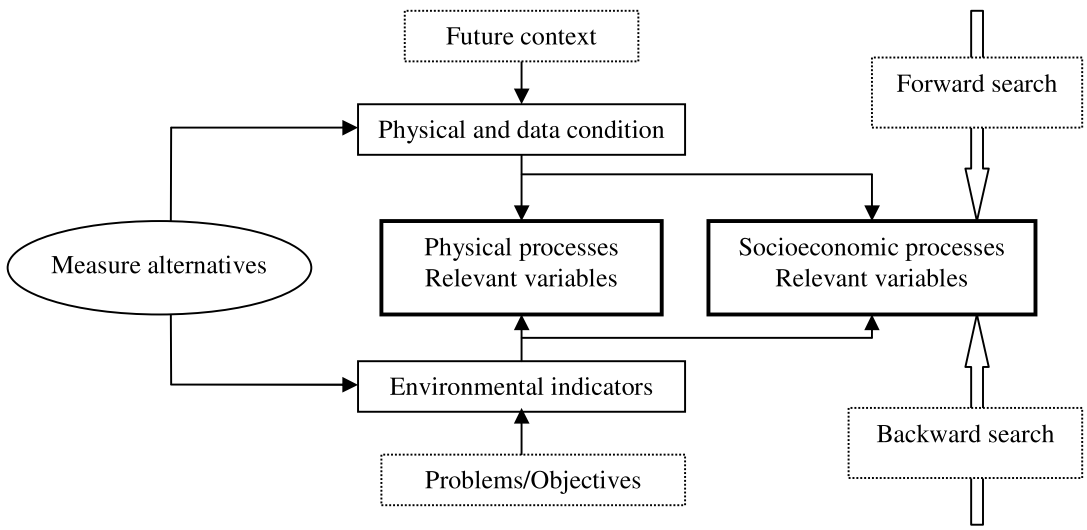

Figure 2.8 *Double-direction search processes followed in this thesis*

As illustrated in Figure 2.8, in *forward* searching, the relevant physical processes are identified through the study of data condition, including future context which can be represented through a change in physical conditions. For example, a hydrological process should be included when climate condition or catchment condition (e.g. land use development) are changed. A hydraulic process should be included when the river geomorphology are changed. The geomorphology changes include land use development that can affect the surface resistance condition, and constructions along/on the river such as dikes and reservoirs, or the deepening or widening a river, which can change the river conveyance condition.

In *backward* causal reasoning, determined by the objectives, the environmental and socio-economic indicators are linked to the measures. For example, for indicator such as the flood mitigation impact indicated as risk or monetary damage, the management measure can be temporal storage in a retention basin, or an intentional dike break at an economically less important upstream area. The consequences of such measures can be simulated with a hydraulic model using environmental indicators such as the water levels along the river, or a socio-economic indicator such as flood damage reduction in monetary terms (obtained by using a risk assessment model).

Benefiting from the definition of indicators, the double-direction search method can lead to a network with more focusing and direct links between system components using output-input relationships. With *double-direction* searching, the design loop can be closed at their meeting point, i.e. the relevant processes and indicators, as shown in Figure 2.8.

2.4.1.3 Formulating a conceptual DSS diagram

The final step of the qualitative phase is to integrate the identified processes and models into a conceptual DSS diagram. During this step, alternative measures are identified and implemented in the conceptual network; processes are connected with each other following their causal links. Measures are linked to the indicators via the processes and are indicated in the diagram. The integration processes and connecting variables are indicated along the links. System outcomes are represented by environmental indicators that are connected with either physical processes, or socioeconomic processes. The completion of the qualitative phase provides a system diagram, and associated model characteristics. Care must be taken that the determination of model complexity is in terms of required model characteristics, which is more than an inventory of available software packages.

2.4.2 Quantitative analysis

Once a conceptual DSS has been set up, the quantitative analysis phase follows Process models are linked through a set of input and output interactions. Activities involved in quantitative analysis are: mathematical formulation, including appropriate numerical solutions when a model is yet to be programmed, modeling package selection and installation. Evaluations of model and system performance are carried out using statistical approaches such as SA and UA. This process should be iterative as well as interactive. The reason is that extensive discussion between modelers and end users about the required functionality and presentation methods can point to problems to be solved, until the design meets the requirements from both science and end users' point of view.

2.4.2.1 Selecting models for a DSS

Physical processes and socio-economic processes are identified in the qualitative analysis phase. The essential activity during the quantitative analysis phase is to describe each process in the form of mathematical equations. When formulating quantitative models, various aspects should be considered: complexity, uncertainty and sensitivity, flexibility, user-friendliness.

Few papers have addressed how to select and formulate a model appropriately (De Kok and Wind, 2002). As discussed previously, the commonly used approach is to compare models among candidates based on their performance. The problem, however, is how to define objective criteria for model selection. Furthermore, such an approach is based on a situation in which there are alternative models, or rather, modeling packages to choose from. Another method for model selection is to use the internal consistency concept (De Kok and Wind, 2002). The idea is to establish connections between processes following causal relationships taking into consideration the physical conditions such as the hydraulic character of the river, soil type, and climate scenarios.

In general, river flows are unsteady. Complexity can only be reduced through elimination of terms in the governing equations, or simplification of the governing equations. For example, steady-non-uniform flow can be used when the discharge does not change along the river. However, once a dynamic indicator such as the velocity change is needed, such a simplification can no longer be applied. Simplification of a hydraulic model can be also desirable for practical reasons, such as avoiding time-demanding computations for larger spatial scales.

In the approach developed in this thesis, the double-direction searching method (Figure 2.8) is applied. Based on the physical conditions such as catchment characteristics, it yields a first classification of models based on the identified complexity in terms of mathematical equations. Once models are formulated in terms of mathematical equations, the next step is to choose the most appropriate model. In general there are two ways to select a model: choose a readily available modeling software package that fulfils the model formulations, or, develop a new model starting at the programming level. The second approach is appropriate when a simple programme is needed. When a complex model is needed, choosing available modeling software package fulfilling the complexity requirement is clearly to be recommended, given the time and effort involved in development and testing.

Choosing an appropriate modeling package can be seen as part of the model selection procedure. Criteria for selection of a modeling software package can be for example functionality, cost, and maintenance. With the introduction of computer technology, mathematical models have been emerging since the 1970s (Abbott, 1991). Various modeling packages were developed during the past 30 years, either free software or commercial software, or inherited from previous research work carried out at institutes. Still, the selection of a modeling software package can be carried out with reference to the model complexity determined during the qualitative analysis phase.

The quantitative design phase provides an integrated system constructed out of different component models through input-output links. To understand model and system performance, as well as to provide comprehensive results to support decision-making, UA of data and models as well as of the integrated system as a whole should be carried out in the quantitative phase.

2.4.2.2 Quantitative model and system analysis

The final step of the system development is the quantitative system and model analysis. First, process models should be evaluated individually, and then the integrated system should be evaluated.

For individual models, the evaluation comprises in general validation and UA. Based on the validity of model aspects as proposed by Finlay and Wilson (1997), and studies of sensitivity and uncertainty on model performance (Snowling and Kramer, 2001), the following criteria can be used to indicate performance:

1. *Flexibility* can be assessed by checking accessibility of parameters in the model.
2. *Data demand* calls for investigating the model requirements in terms of available data.
3. *Model accuracy* is to evaluate the quality of the output, and the way that model results should be interpreted.
4. *Sensitivity* is defined as the amount of change in model output resulting from a change in model input (Morgan and Henrion, 1990). Both individual and global sensitivity analysis is required to determine sensitivity of the model.

5. *Uncertainty propagation* or *error propagation* is often carried out by using a Monte Carlo simulation (Morgan and Henrion, 1990; Saltelli et al., 2000). However, UA is often only carried out for individual models; no report has been found on UA in integrated systems involving several models with a hierarchical structure.

2.4.3 Communication

Communication between the modelers and decision-makers or end users, plays a vital role in the development of a DSS. The final goal of developing a DSS is to provide end users, or decision-makers, with comprehensive and correct information to facilitate decision-making. Therefore, communication is essential not only for system improvement such as comprehensive and understandable result presentations, but also for knowledge distribution and education.

Interactions occur not only between modelers and decision-makers (or end users), but also between modelers who are working in different fields, or at different stages of the design. For parallel development of models, communication is mainly about the inherent relationship and integration of these models, i.e., the linking of models.

2.5 Uncertainty Analysis

Uncertainty analysis and sensitivity analysis are found to be the most appropriate techniques to evaluate model and system performance. According to Morgan and Henrion (1990), a distinction can be made between:

- Methods for computing the effect of changes in inputs on model predictions, i.e., *sensitivity analysis*;
- Methods for calculating the uncertainty in the model output induced by the uncertainties in its inputs, i.e., *uncertainty propagation*; and
- Methods for comparing the importance of input uncertainties in terms of their relative contributions to uncertainty in the outputs, i.e., *uncertainty analysis*.

Some literature treats SA as part of UA (Saltelli et al., 2000), as it utilizes sampling-based approaches for the analysis. In practice, SA provides the significance of the contribution of each variable to the uncertainty, where UA is used to determine how well the model performs.

Nevertheless, to obtain the uncertainty in the indicators, the following steps should be taken: uncertainties identification, SA, uncertainty propagation through the model, and uncertainty propagation through the integrated system. The following sections describe these steps and methods applied in this thesis in more detail.

2.5.1 Uncertainty identification

The first step of UA is to identify uncertainty sources. This step is particularly appropriate for state variables related to the environment. Based on their sources, uncertainty can be classified as (Morgan and Henrion, 1990):

- Random error and statistical variation – direct measurement errors, occurring in data and parameters (Quantification can be in terms of standard deviation, confidence intervals, among others);
- Systematic error and subjective judgment – the difference between the 'true' value of the quantity of interest and the value to which the mean of the measurements converges as

more measures are taken (analysis of a systematic error may provide a range or error bounds);

- Linguistic imprecision – people's understanding and interpretation of the outcomes of a system;
- Variability – uncertainty about the probability distributions a parameter can adopt;
- Inherent randomness – natural variations within real-world quantities (environment);
- Disagreement – different scientific points of view from experts;
- Approximation – assumption needed because a model is a simplified version of the real word.

These sources of uncertainty should be analyzed before uncertainty propagation is carried out. Quantification of uncertainty includes the definition of its statistical distribution and range of variations. Quantification of uncertainties is normally carried out by data analysis and literature study. For parameters for which quantification is not immediately possible, interviewing experts might be of help to define the range of likely variations.

2.5.2 Sensitivity analysis

Sensitivity analysis is the study of how the variation in the output of a model (numerical or otherwise) can be attributed, qualitatively or quantitatively, to different sources of variation in information fed into it (Saltelli et al., 2001). By definition, the sensitivity index S_i can be calculated using the normalized value determined from the mean of the output y^0 and the mean of the input x_i^0:

$$S_i = \frac{\partial y}{\partial x_i}\frac{x_i^0}{y^0} \qquad (2.2)$$

Alternatively, the sensitivity index could measure the effect on y of perturbing x_i by a fixed fraction of x_i's standard deviation, i.e.

$$S_i = \frac{\partial y}{\partial x_i}\frac{\sigma(x_i)}{\sigma(y)} \qquad (2.3)$$

where σ indicates the standard deviation of the parameter variations.

SA has proven to be an appropriate technique to facilitate effective model calibration. Through SA, the significance of model output changes associated with input changes is obtained. With the help of SA, a modeler can focus on the calibration of the parameters the model is most sensitive to.

There are different approaches to assessing the sensitivity of model parameters. *Local* SA concentrates on the local impact of each parameter in the model. It is usually carried out by computing partial derivatives of the output functions with respect to the input variables. It is in fact a particular case of the One-At-a-Time (OAT) approach, since when one factor is varied, all others are held constant at their central (nominal) values. However local SA is less helpful when SA is used to compare the effect of various factors on the output, as in this case the relative uncertainty of each input should be weighted. In contrast, a *global* SA evaluates the effect of x_i while all other x_j, $j \neq i$, are varied as well. In doing so, the global SA incorporates the influence of the whole range of variations and the form of the probability density function of the input. Therefore, it assigns the uncertainty in the output variable to the uncertainty in each input factor. To carry out a global SA, a sampling-based method is required to generate a representative uncertainty domain of parameters and variables.

There are many quantitative SA techniques. For local SA, the relative variation of y due to perturbing x_j by a fixed fraction of x_j's central (or nominal) value, the sensitivity density (or sensitivity index) can be used, expressed as:

$$\frac{\partial y}{\partial k_j} \approx \frac{y(k_j + \Delta k_j) - y(k_j)}{\Delta k_j}, j = 1, \ldots, m. \qquad (2.4)$$

where y is the model output, Δk_j is the variation of input variable x_j. To normalize the indicator the variations are proportional, i.e. in % of a nominal value. For global SA (sample-based analysis), the computation time of a model is a major problem. Studies show that Morris' method is a rather effective and efficient method particularly when large numbers of parameters are involved (Morris, 1991).

Morris's idea is to calculate the elementary effects by using the OAT approach. The method belongs to the simplest class of screening designs (Saltelli et al., 2001) for SA. The standard OAT designs use the 'nominal' or 'standard' value per factor. The combination of nominal values of the k factors is called the 'control' scenario. The importance of each input is obtained from the difference between the outputs for the extreme inputs and the standard factor values. The original OAT design is a local SA approach, which is acceptable only if the input-output relationship can be adequately approximated through a first-order polynomial. If the model shows strong nonlinearity, then Morris's OAT method is more preferable due to its ability to cover the entire space over which the factor may vary.

With Morris's OAT, the k-dimensional factor vector X for the simulation model has components x_i that have p values in the set $\{0, 1/(p-1), 2/(p-1), \ldots, 1\}$. The region of experimentation Ω then is a k-dimensional p-level grid. In practice, the values sampled in Ω are subsequently rescaled to generate the actual (non-standardized) values of the simulation factors. Let Δ be a predetermined multiple of $1/(p-1)$; then Morris defines the *elementary effect* of the i^{th} factor at a given point X as:

$$d_i(X) = \frac{[y(x_1, \ldots, x_{i-1}, x_i + \Delta, x_{i+1}, \ldots, x_k - y(X)]}{\Delta} \qquad (2.5)$$

To get the elementary effect, Morris developed an economical design, which requires $r \times k$ runs in total, where r is the number of levels representing how many elements (at equal intervals) the sampling takes. To apply Morris's method, the following steps should be followed (Saltelli et al., 2001): 1) A 'base' value X^* is randomly chosen for the vector X, each component x_i being sampled from the set $\{0, \frac{1}{p-1}, \ldots, 1-\Delta\}$. One or more of the k components of X^* are increased by Δ such that a vector (say) $X^{(1)}$ results that is still in Ω. The estimated elementary effect of the i^{th} component of $X^{(1)}$ (if the i^{th} component of $X^{(1)}$ has been changed by Δ) is

$$d_i(X^{(1)}) = \frac{[y(x_1^{(1)},...,x_{i-1}^{(1)}, x_i + \Delta, x_i^{(1)},...,x_k^{(1)} - y(X^{(1)})]}{\Delta} \quad (2.6)$$

where change step Δ can be either positive or negative. Let $X^{(2)}$ be the new vector defined in the above step. Select a third vector $X^{(3)}$ such that $X^{(3)}$ differs from $X^{(2)}$ for only one component *j*. The estimated elementary effect of factor *j* is then

$$d_j(X^{(2)}) = \frac{y(X^{(3)}) - y(x^{(2)})}{\Delta} \quad (2.7)$$

Repeat this step such that a succession of $k+1$ input vectors $x^{(1)}$, $x^{(2)}$, …, $x^{(k+1)}$ is produced with two consecutive vectors differing in only one component. Furthermore, any component *i* of the 'base vector' X^* is selected at last once to be increased by Δ, to estimate one elementary effect for each factor.

The global SA and Morris's OAT method have been found to be more effective and were applied in this research.

2.5.3 Uncertainty propagation through individual models

Uncertainty propagation through individual models (uncertainty transformed from input, through the model, to the model output) can be carried out using Monte Carlo simulation (MCS). MCS is based on multiple evaluations with randomly selected model input, and then using the results of these evaluations to determine both uncertainty in model simulations and their contribution to the overall uncertainty range (Morgan and Henrion, 1990; Saltelli et al., 2000). In general, the analysis involves the following steps:

- Selection of ranges and parameter distributions for each input X_i;
- Generation of a sample from the ranges and distributions specified in the step 1;
- Running the model for each sample;
- Carrying out uncertainty analysis.

Parameter distributions are obtained from the literature and/or empirical knowledge. For example, the Gumbel distribution (Shaw, 1994) is used for the annual maximum discharge, a uniform distribution is adopted for most of the parameters with unknown distributions.

Sampling methods is a statistical approach to select multiple combinatorial input scenarios from the distributions. A sample is a set of parameters used for running the model to obtain a set of outputs. The simplest sampling method is the *Random sampling*, i.e. Monte Carlo analysis. A more efficient method is the *stratified sampling* (Morgan and Henrion, 1990). The most commonly used form of stratified sampling method is *Latin Hypercube Sampling* (LHS) (Mckay et al., 1979). LHS performs better than the previous two sampling strategies when the output is dominated by only a few components of the input factors. The method ensures that each of these components is represented in a fully stratified manner, no matter which component might turn out to be important, and therefore gives a more stable estimate of the mean.

Sample size is the number of combinations of parameters. Selection of the sample size depends both on the cost of each model run, and what one wants the results for. Morgan and Henrion (1990) provided a method to estimate sample size assuming primary interest is in the precision of the mean of the output variable *y*. To obtain a α confidence interval smaller than one ϖ unit wide, a small Monte Carlo run is needed to get an initial estimate s^2, the variance. The sample size m can be estimated as:

$$m \geq (\frac{2cs}{\varpi})^2 \tag{2.8}$$

where c is the deviation for the unit normal enclosing probability α .

Uncertainty analysis is to determine the uncertainty of parameters and variables expressed using probability concepts, such as the expected value and variance of the output variables estimated by:

$$\hat{E}(Y) = \frac{1}{N}\sum_{i=1}^{N} y_i \tag{2.9}$$

$$\hat{V}(Y) = \frac{1}{N-1}\sum_{i=1}^{N}[y_i - \hat{E}(Y)]^2 \tag{2.10}$$

while sample-based UA is applied.

2.5.4 Uncertainty propagation through integrated systems

Methods for UA involving multiple models are not explicitly addressed in the literature. A stepwise sample-based uncertainty can be applied, i.e., for the input model, UA is carried out to obtain a set of output variables. Uncertainty of the output variables can be considered in the sampling of the next model that utilizes those variables as inputs. Computational runs of such a stepwise approach are the sum of runs of each individual model.

The stepwise method can only be applied when models do not require excessive computational loads. For models such as a 2D hydraulic model with a large spatial domain and number of time steps, stepwise sample-based approaches are difficult to apply. To reduce the computational load, the scenario tree method (Morgan and Henrion, 1990) is adopted, which gives a set of combinatorial scenarios for UA (Figure 2.9).

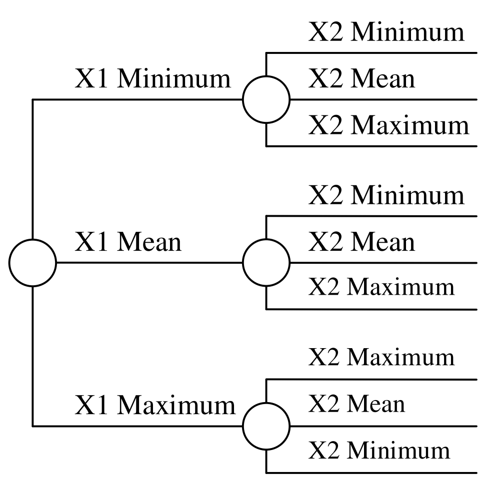

Figure 2.9 *Example of a scenario tree that uses three quantitative levels*

As shown in Figure 2.9, each node represents an uncertainty quantity or event, and each branch from that node, one of its possible outcome values. Each path through the tree from root to top represents a sequence of event outcomes determining a specific scenario. The combinatorial scenarios define an uncertainty domain, which can be propagated to the next model(s).

2.5.5 Outcome presentation with uncertainty

The sampled model output and corresponding distribution can be used to distinguish scenarios, or alternative measures or models (Figure 2.10). For example, the scatter diagram in Figure 2.10a indicates an obvious difference in the costs of scenarios I and II. Figure 2.10 shows that the costs of scenario I are higher than the costs of scenario II, which leads to a preference for the implementation of scenario II.

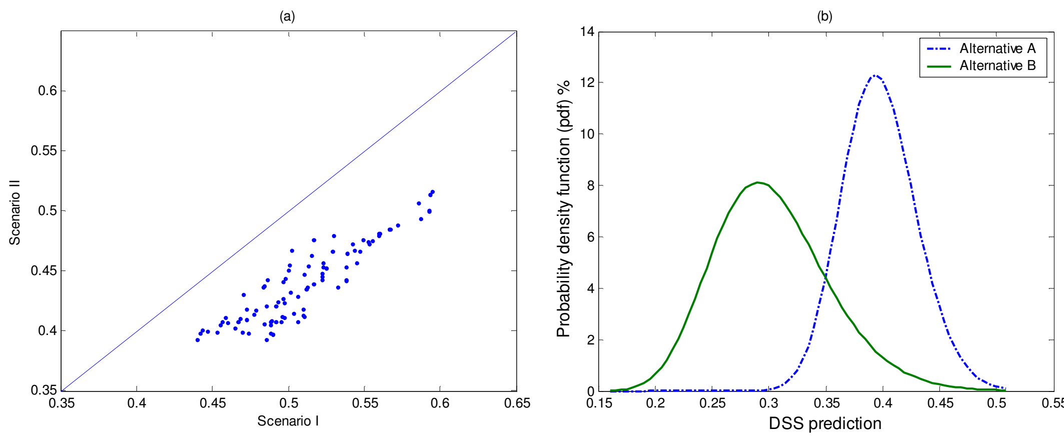

Figure 2.10 *Examples of uncertainty analysis: (a) scatter plot of sampled output results; (b) probability density function of alternative measures*

A similar comparison can be made using another presentation method, such as the probability density function fitted for different measures (De Kok and Wind, 2003) (Figure 2.10b). This method plays a vital role in the implementation stage of a DSS. For example, a higher mean value and a smaller variation of *Alternative A* can be found when comparing with *Alternative B* and using the probability distribution plot as shown in Figure 2.10b. This information can be used to judge the appropriateness of implementation of measures according to the desired DSS prediction when the difference is distinguishable. When there is too much overlap to be able to distinguish between the two, the scatter plot (Figure 2.10a) can be of additional help (Huang et al., 2005).

Such a comparison can benefit model selection as well. For example, a model that can show clear differences among different scenarios or implementation of measures can be considered more preferable than a model that cannot show such distinctions. For this purpose, it is more reasonable to compare the result of implementation of a measure relative to a nominal situation, i.e. no implementation of any measures. The reason is that the results are not always distinguishable when comparing consequences of different measures, which often happens at the implementation stage of a DSS. For example, in terms of effects on the conveyance of a river, deepening the river may provide the same effect as widening the river, which may result in similar distribution curves for both measures; this, however, does not mean that the hydraulic model is not appropriate for the purpose it serves.

2.6 Summary

In this Chapter, the state of the art of DSS design and FRA has been reviewed. Current design approaches for DSS for IRBM have been found to be unsatisfactory due to the lack of a method for model selection, which may lead to excessively complex or overly coarse models and system design. Another issue is the lack of a system performance evaluation method, which is required to assess the usefulness of a DSS during its development period, and provides more comprehensive impact of consequence for the implementation of measures during its implementation period.

The history of FRA is briefly reviewed in this Chapter. Two types of risk assessment approaches have been found throughout the history of FRA, namely the statistical approach and the physically-based approach. Using the risk-analysis approach, the statistical approach aims to provide a statistic risk distribution with the outcome of expected annual damage, which is often used for long-term planning, whereas the physically-based approach calculates the damage/risk associated with a certain flooding event by using hydrodynamic computations, which is found to be more suitable for short-term operational activities such as evacuation. More specific problems are found in the development of IFRA, particularly on the quantitative inclusion of flow velocity (which means that FRA is incomplete) as well as difficulties in the selection of hydraulic models.

The concept of appropriate modeling leads to an approach based on system analysis for the design of DSS for IRBM. The approach comprises two interactive and iterative phases, namely the qualitative analysis and quantitative analysis. Qualitative analysis tries to achieve a conceptual system framework, and identifies model and system complexity based on scientific principles to do that. The quantitative analysis is focused on an operational DSS, and to that end, formulates models with mathematical equations and integrates the system with models and data. Other activities included in this phase are model calibration and validation, and UA.

Aiming at an appropriate formulation and selection of models, a *double-direction searching* method is proposed (Huang et al., 2005) for the phase of qualitative analysis. This method optimizes the system diagram by increasing the focus on model and variable selection incorporating the concepts of internal consistency (De Kok and Wind, 2002) and the definition of environmental and socio-economic indicators (EEA, 1999). With forward searching, the method identifies the relevant processes and parameters following the information flow, and is constrained to the backward searching which traces management indicators back to relevant measures and processes by causal reasoning.

UA is found a key procedure for model and system performance evaluation. It has been also found useful to distinguish impact of different measures. Approaches of sensitive analysis including global SA and Morris's One-at-a-Time approach are presented and adopted later in this thesis (Chapter 5). Uncertainty propagation through sample-based Monte Carlo simulation with a Latin Hypercube Sampling approach is also introduced. To save large computation loads and time, a scenario tree is employed for UA in full 2D hydrodynamic modeling.

Chapter 3

Conceptual Framework for Integrated Flood Risk Assessment - The Elbe Case Study

Building on the guidelines for model selection and different risk assessment approaches discussed in chapter 2, and the experience gained during the development of a pilot DSS for the Elbe River, a qualitative/conceptual framework for IFRA is proposed in this chapter.

In 2001 the German Federal Institute of Hydrology initiated a project aimed at developing a prototype DSS for integrated management of the Elbe river basin, focusing on the functions of navigation, vegetation ecology, flood safety, and water quality (Matthies et al., 2003). After the August 2002 flood catastrophe flood risk became a major issue, and the design of the prototype was redirected to include flood risk indicators at several scale levels, with different purposes.

This chapter introduces the physiography, hydrology and flood defense system of the Elbe River. Issues required to be studied in the pilot Elbe_DSS such as the problems related to flood risk, management objectives and corresponding indicators, are discussed together with the potential measures for flood mitigation available to the authorities. The results are applied with the double-direction search algorithm proposed in Chapter 2 to arrive at a qualitative framework for FRA. Depending on the nature of the problem and model as well as data conditions, the framework can be used to choose between a statistical and a physically-based risk assessment approach. In the physically-based approach, attention is paid to the generation of an extreme flood event as the upstream boundary for hydrodynamic modeling of actual flooding.

To incorporate the velocity effect, a risk matrix is introduced, which is an important process of the framework. The principle is a symmetric classification of risk level associated with the combination of inundation damage and damage caused by velocity, which results in a risk level map. Each risk level corresponds to a flood management activity. The risk levels in the matrix are to be defined by decision makers and subjective, but can be a useful instrument for long-term planning.

3.1 Introduction

Sustainable flood management requires insight in the consequences of combinations of implemented measures, such as dike shifting, at locations of interest along the Elbe River, taking into account changes in the future context such as climate change and land use development. This can be done through a comprehensive FRA of the study area.

The two most important aspects of flood risk are the flooding probability and the flood damage. There have been many ways of dealing with flood risk. For example, in the Netherlands in the past people tended to live on higher grounds. In this way flood damage was as low as possible. Damage reduction was the major concern. In the course of time the higher grounds were connected by dikes to protect the land from flooding. Reducing the probability of flooding became the leading strategy (Parmet, 2003). However, due to rapid climate change and economic development, dealing with flood risk is more than maintaining safety standards for flood defense systems (Dijkman et al., 2003). This leads to a shift in policy priority from 'structural measures' to 'non-structural measures', such as the shifting from increasing dike height to the using of retention basins. Choosing the right measures, either structural or non-structural, in cooperation with experts in social and engineering sciences, is more and more recognized to rely on appropriate assessment of the flood risk.

As summarized by Vrouwenvelder et al. (2003), major steps in FRA include:

1. Identification of undesired events and scenarios;
2. Assessment of the relevance of the scenarios', the probability of occurrence of the scenarios'; and
3. Assessment and quantification of the consequences of the flooding event.

The *first* step is to recognize and qualitatively describe a hazardous situation. Aspects relevant for flood damage can be categorized as basic data including a elevation and land use data; hydraulic characteristics such as the water depth, flow velocity and flooding duration, sediment concentration/size, wave or wind action, pollution load of flood water, rate of water rise during flood onset, and socio-economic factors such as awareness and warning/response variables (Van der Sande, 2001). These factors depend on the nature of the event such as heavy rains or storms, high river discharge, high sea level, including human errors such as ships hitting flood defense structures. Floods may cause further events such as a dike breach, or overtopping of the dikes. In addition, indirect hazards after the floods should also be taken into consideration, for example the presence of chemical plants or spread of diseases. Qualitative identification of those hazards is important to search for appropriate measures to prevent such effects.

The *second* step is a quantification of the probability of occurrence of a flood. Flood protection engineering starts with the identification of the locations where a defense system may fail, as well as the responsible factors. Then, on the basis of models and statistics for the hydraulic loads and strengths of the system, one calculates the event probability. For example for the overtopping of a dike is associated with the design probability of a dike height.

The *third* step is the modeling of consequences. It starts with the calculation of the water level and flow velocity in the flooding area. Based on land use data, three typical consequence can be distinguished as: *direct damage* such as damage due to the direct loss of means (financial damage expressed in monetary terms; number of *people* who are affected/injured) and recovery damage to recourses in possession or rent; and *indirect damage* due to, for example, business interruption, environmental damage, cleaning costs and evacuation costs.

For decision making purpose additional analysis is required to:

1. Assess the acceptability of the risk(s) determined; and
2. Identify and assess the effectiveness of risk reduction measures.

Risk acceptance is based on various criteria such as laws or regulations, standards, experience, and/or theoretical knowledge about the acceptable risk. The criteria for acceptance may be expressed verbally or numerically, and should be considered as a matter of debate. Once established, laws and habits need to be updated from time to time due to a continuous change of circumstance and insights. When the flood risk is considered too large for direct acceptance, one should look for adequate risk reduction measures.

Risk reduction measures are aimed at preventing a flood event from occurring or at mitigating the consequences (Zhou, 1995). Examples of preventive measures are the design and maintenance of flood defense systems (dikes, storm surge barriers, locks and sluices, river works), temporal water storage in upstream areas, or the use of a retention basin. To mitigate the flood risk, measures can be evacuation shortly before the flood, and/or rescue operations taking place in case of actual flooding.

However, the three steps of the assessing of flood risk are associated with a certain flood event. This does not include a general risk assessment for the river basin in a catchments scale. As reviewed in Chapter 2, the statistical risk-analysis approach provides expected annual damage (Stedinger, 1997) can be used to provide a static flood risk distribution in relation to physical and economic (land use) conditions, in addition to the common risk assessment described previously. This approach is also adapted in the IFRA presented in this chapter.

Following the steps described above, for the development of Elbe_DSS, a conceptual IFRA for flood management has been established using the appropriate modeling concepts introduced in Chapter 2.4. The developing procedure consists of:

1. Study of physical conditions of Elbe River (section 3.2), including geographical and hydrological condition, which reflects the forward searching process included in the double-direction search algorithm (Chapter 2, Figure 2.8);
2. The backward searching procedure is applied in order to identify flooding problems and management activities for Elbe River, as well as environmental and socio-economic indicators (section 3.3);
3. Based on the analysis of physical conditions and problem/objective identification, incorporating the specific principles (equations and parameters requirements) for each flood risk assessment approach, *section 3.4* identifies the relevant processes and parameters for the statistical risk assessment approach and the physically-based risk assessment approach. According to the requirement from processes and parameters, the objective indicators and risk assessment approach, as well as the data condition, hydraulic models are selected (Figure 3.7, Figure 3.8). In order to incorporate the effect of flow velocity, the use of a risk matrix is proposed. The above processes have resulted in an appropriate model/system complexity at a conceptual level, as illustrated in Figure 3.9.

3.2 The river Elbe

The River Elbe (ĕl´be), Czech *Labe,* is one of the longest rivers of central Europe. It is 1,170 km in length, of which 700 km is in Germany (Figure 3.1, next page), and the complete river-basin area is 148,268 km^2. It rises in the Krkonoše (Giant) Mountains on the border of the Czech Republic and Poland and flows southwest across Bohemia. It then flows northwest across Germany and empties into the North Sea near Cuxhaven. It flows generally NW through Eastern Germany (past Dresden, Wittenberg, and Magdeburg) and onto the North German plain. The main tributaries of the Elbe are the Vltava, Mulde, Saale, and Havel rivers. A canal system connects the Elbe with the Berlin canal system and the Oder River (to the east), as well as with the Ruhr region, and the Weser and Rhine rivers to the west; and with the Baltic Sea to the north.

The territory of four countries is included within the catchment basin of the Elbe River. A total of almost one percent of the Elbe catchment lies in the countries of Austria and Poland. 99% of Elbe river basin is confined to only two: the Czech Republic (*1/3*) and Germany (*2/3*).

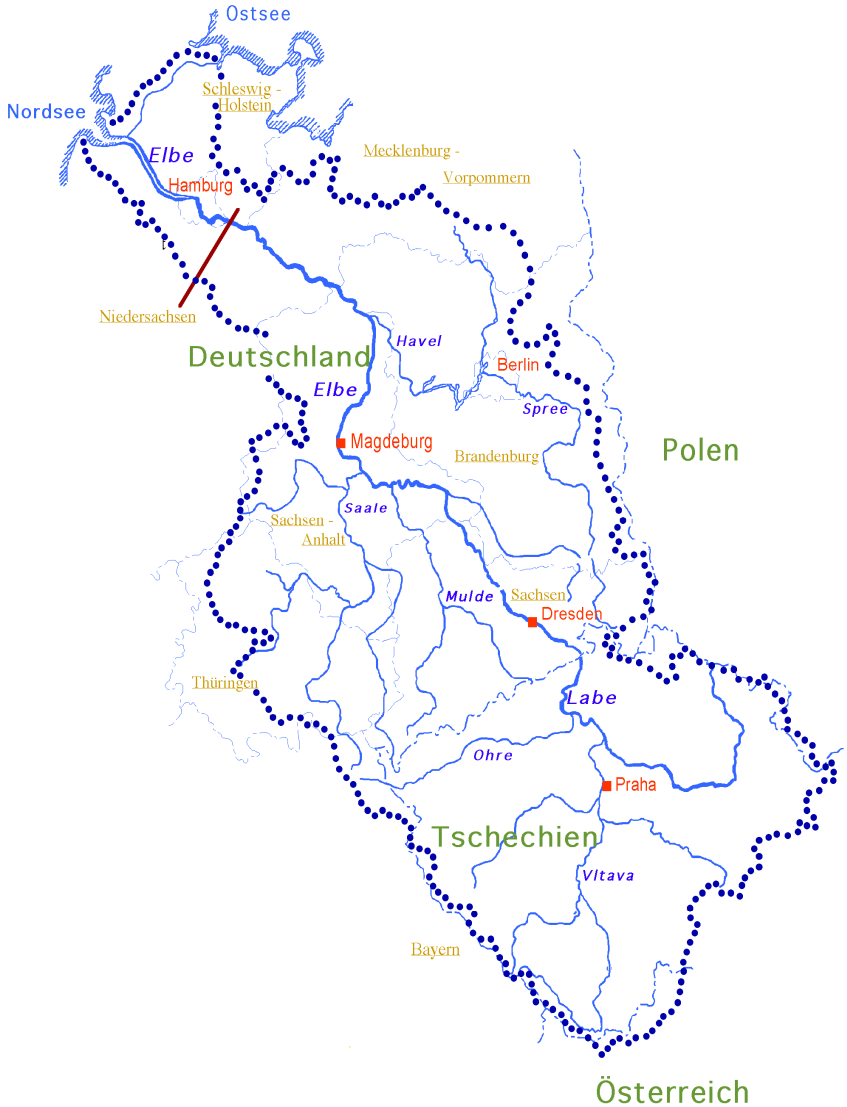

Figure 3.1 *The Elbe River basin (ELBis, 1996)*

3.2.1 Geographic conditions

Three main landscape types can be distinguished along the upper, middle and lower reaches of the Elbe. In its high altitude origin of the Czech Republic, where it is called the Labe, tall peaks and cliffs of weathered sandstone dominate the region. The main historical city of this region is Prague in Czech Republic, which lies on the Vltava, the longest river in the Czech Republic, not far before its Elbe junction. The population density of the area in general is not very high, with small industrial sites and agricultural settlements predominating. The presence of many built structures also impacts the character of the landscape within the upper reach of the Elbe.

The middle stretch of the River Elbe is characterized by expanses of relatively flat landscape. Here too, agriculture plays a large role and industrial locations are prevalent; however, engineered structures to control the waterway are restricted to tributaries and largely absent along the Elbe. The population density is higher, with the historical city of Dresden located at km 200 along Elbe starting from the border between Germany and Czech Republic, and Leipzig located approximately 120 km away from the river Elbe, along the White Elster, a tributary of the Saale, a main tributary of the Elbe. The most notable feature of the middle Elbe landscape is the UNESCO biosphere reserve, which incorporates 3,744 km^2 of continuous, largely original floodplain forests. Within the lower middle reach of the Elbe, the extensively canalized and controlled tributary, the Havel, joins the Elbe.

The lower reach of the Elbe River in Germany is formed by the beginnings of the estuary and coastally-influenced landscape. The harbor city of Hamburg and especially its container shipping port dominate the structure of the river during much of this lower stretch. Tidal fluctuations lead to very special forms of wetland habitat and biodiversity. For the population and industry located in this region, storm surges are a recurring phenomenon.

In the following sections, chainage of the river is counted as zero from the border between Germany and Czech Republic.

3.2.2 Hydrological characteristics

The longitudinal annual average discharge of Elbe varies from 1300–1900 m^3/s along the river length from Dresden (km 55.6) to Neudarchau (km 536.5), discharge of 100 year return period varies from 2700 – 4000 km^3/s, as shown in Figure 3.2 (Helms et al., 2002).

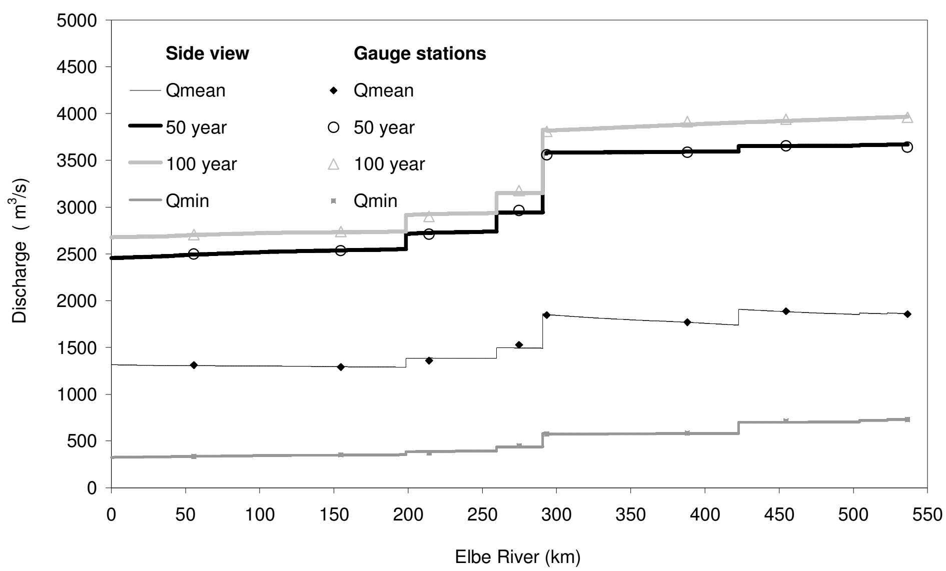

Figure 3.2 *Side view of discharge distribution along Elbe River in Germany*

In addition to the flow from upstream in Czech republic as well as flow along the main channel river basin, there are four major tributaries contribute discharge for Elbe river in the basin of German part, these are the Schwarze Elster (km 195), the Mulde (km 259), the Saale (km 290) and the Havel (km 445), contributing 5%, 7%, 19% and 17%, for the main channel flow, respectively (Figure 3.3).

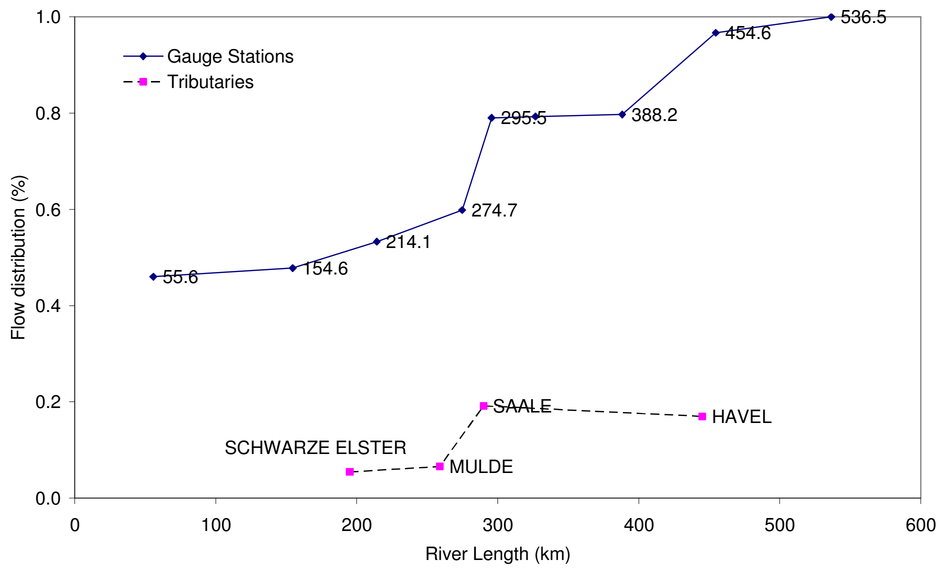

Figure 3.3 *Flow distributions at gauge stations along Elbe, including contribution from tributaries*

There are 35 years (1960-1995) time series of daily discharge and water level at gauge stations along main channel of Elbe as well as gauge stations at major tributaries (Helms et al., 2002a,b) (Figure 3.4)

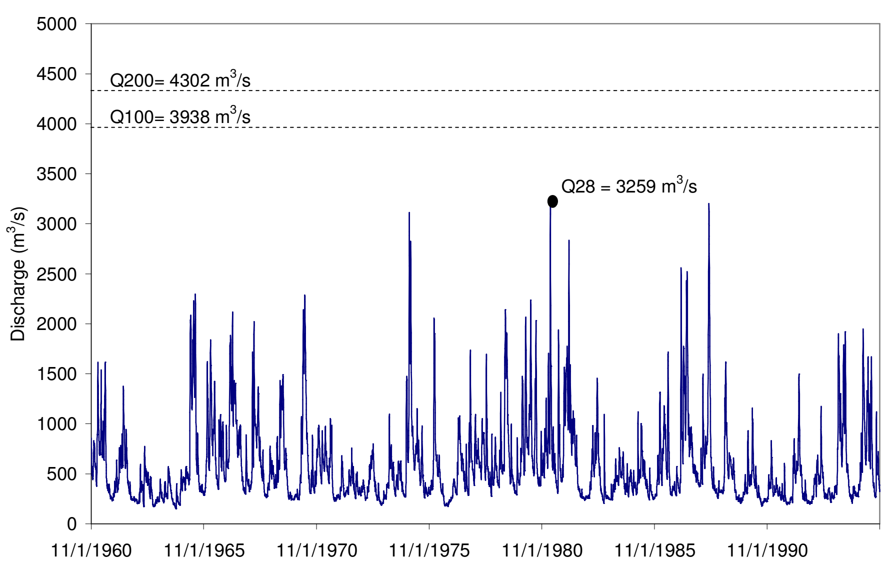

Figure 3.4 *Daily discharge of 1960-1995 at gauge station Tangermünde (km388.2)*

As shown in Figure 3.4, the maximum year-extreme discharge in the period of 1960-1995 has return period of 28 years. Statistical parameters such as the flood exceedance probability coefficients are obtained using this time series. This means extrapolation is needed to estimate higher return period discharge. The yearly extreme discharge can be described by means of a Gumbel distribution (Shaw, 1994).

Applying the estimation, a Gumbel distribution is fitted using the 35 years time series. As mentioned earlier, the discharge data for the period 1960-1995 are limited for year-extreme discharges values, extrapolation is needed (Figure 3.5). Thus, the uncertainty introduced by extrapolation should be considered.

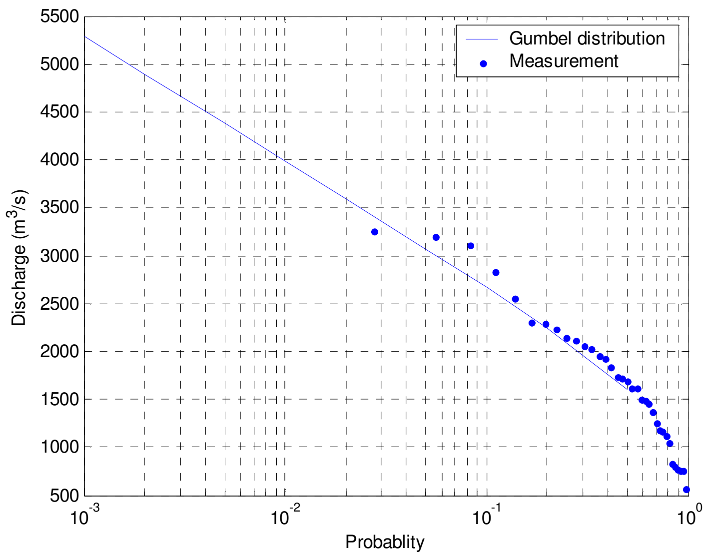

Figure 3.5 *Fitted Gumbel distribution for extreme discharge value at station Tangermünde*

3.2.3 Flood defence system - dikes

The most important flood defense system along Elbe is dikes. Human settlement along the course of the Elbe has a long history, and original attempts to build dikes along its shores to promote fishing and mining date back to around 900 AD (Stubbs and Bonjour, 2002). Early flood protection dikes were already being built in the 1100's. While the Elbe probably retained most of its ecologically naturalistic form and condition in the 1700's. To protect people from flood hazard, numerous dikes have been built in the Elbe River basin. Known as the Albis to the Romans, the river marked the farthest Roman advance into Germany (9 BC) and was later the eastern limit of Charlemagne's conquests. The Treaty of Versailles internationalized its course from the Vltava River to the sea in 1919, but Germany disputed its internationalization after the Munich Pact in 1938. In 1945 the river was made part of the demarcation line between East and West Germany.

Dikes are one of the most sensitive factors for flood risk (De Blois and Wind, 1996). The dikes along the main channel of the German have been constructed with different return period for overtopping (IKSE, 2001). The return period is a statistical concept, expressing the average time interval in years between flood levels reaching the dike crest. Figure 3.6 shows the flood protections level of the dikes along the Elbe River between Tangermünde and the weir at Geesthacht. The return period associated with the dikes in this area varies from 10 years to 100 years, without additional height of safety board.

Distinguished for the left-hand and right-hand side, dike data includes the location of the dike, formatted as polylines GIS shape files, as well as a table with the dike height identified with river kilometer value. The geographical information is carried by the dike line as a GIS shape file, which came originally as discontinuous polylines.

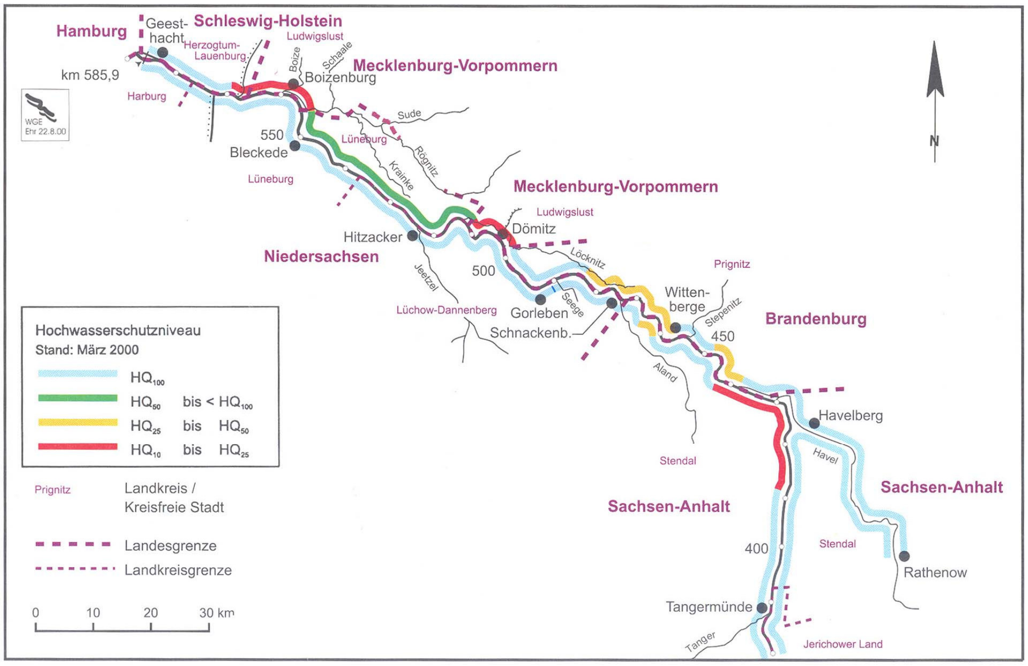

Figure 3.6 *Dikes along the Elbe River between Tangermünde (km 388.2) and Geesthacht (km 585.9), including tributary Havel (IKSE, 2001)*

Complete dike data including the height and location are essential for volume-based modeling of the flood risk. Dike height is a key input variable and criterion for the occurrence of overtopping for a one-dimensional hydraulic approach. Completeness of the dike data is also important to distinguish the embankment area in two-dimensional overland flow modeling for a dike break simulation. During the preparation of the simulations for the testing of the IFRA (Chapter 4), it turned out that the complete dike data were not yet available. In some locations the geographical situation was such that dikes were absent (Jankiewicz et al., 2005). For instance, for the mountain area upstream of city of Dresden there was no need to construct a dike. However, in view of the requirements for hydraulic computations, the original dike line could be considered "incomplete" in some parts due to the gaps appeared in the dike lines. Therefore the dike lines were manually completed by linking the missing gaps following the river shape and the topographical maps, with the exception of the tributaries.

A source of uncertainty of the completed dike line is the miss-shaping due to the re-projection of the dike line from its original projection method into the DSS projection of *Lambert conformal conic QGS 1984* (Pearson, 1984). As most of the GIS data were collected from different sources, and had to be re-projected into a uniform projection system, the re-projection caused a small mismatch between the dike data and other data. For example, it has been found that the dike lines cross the river in some locations. Those uncertainties shall be taken into consideration particularly when the dike data are used for a 2D hydraulic approach.

3.3 FRA for Elbe_DSS

Due to the increasing demand from society on the use and protection of water bodies, and strategic policy making, the German Federal Institute for Hydrology initiated a project in 2000 to develop a generic Decision Support System (DSS) for sustainable management of the Elbe River Basin (Matthies et al., 2003). This tool is based on the interdisciplinary coupling of models and data collected in a research program funded by the German Federal Ministry of Education and Research (BMBF) (Gruber and Kofalk, 2001). Functions addressed by the Elbe prototype DSS are: shipping, water quality, flood safety, and the floodplain vegetation, in various spatial and temporal scales. As one of the important functionality for flood management, FRA is required on both the river basin scale (entire river basin in the Germany part) and the local scale (size of 10-50 kilometers along the river reach).

As the major steps of the quantitative study, the following issues are discussed in the following sections:

1. Problems of flood risk management along the Elbe;
2. Definition of management objectives and features of the DSS;
3. Identification of suitable measures for flood risk mitigation;
4. Defining scenarios for the future contexts (or scenarios);
5. Identification of indicators of FRA that are of particular interest for the Elbe_DSS.

3.3.1 Flooding in Elbe – problem definition

In August 2002, a large flood, with a 200-year return period at the city of Dresden, occurred in Middle Europe. The monetary loss was estimated to be 9.2 billion euro in Germany and 3 billion euro in the Czech Republic (Schanze and Reinke, 2003). Precipitation reached 300 mm per day with peaks of 25 to 30 mm per hour in the Ore Mountains. It has been widely discussed and accepted that due to the rapid climate change the increased temperatures of the atmosphere will provide a more frequent recurrence of extreme precipitation and floods. Possibly, this “flood of the century” could repeat itself more frequently in the near future in the Elbe catchment.

Different causes have been identified for flood damage in general, as well as for the flood of 2002. In 2002, two types of floods occurred: rapid flash floods of the Elbe tributaries from mountainous catchments with high velocity, sediment transport, and slow floods at the lower parts of the river reach with a deeper and broader inundation pattern. The two types of floods differ fundamentally in terms of the damage and underlying causes.

In the mountains collected data on weather prediction, reservoir management and land use in the floodplain areas were most meaningful. The day before the event precipitation had been estimated to be 40-80 mm. Thus the water management authorities and public were less alerted. Because of the summer season the reservoirs were filled for water supply up to the fixed flood prevention level. In the densely settled valleys buildings and infrastructure were not prepared for the extreme event that followed. In the lowlands the reduction of flood plains and detentions areas, the flood protection level and settlements in the flood prone areas caused very high water levels, high inundation depths, resulting in high flood damage. Especially after the political reunification economical pressures had softened the strict protection of the flood zone. Therefore, clearly, to reduce flood risk for those important areas, or areas of significant interests, effective flood mitigation measures needed to be implemented (DKKV publication, 2003).

The extreme event pointed to a need for a policy shift to non-structural measures. Previously, flood mitigation measures were based on the assumption of a desired level of flood protection. In terms of extreme events limitations to the level of flood protection that can be achieved must be accepted. To deal with flood risks instead of focusing on the avoidable natural hazards, non-structural-measures such as an intentional artificial dike break, or the use of a temporary storage, or shifting the dikes, are needed. It is clear that apart from requirement of measures involving different spatial and temporal scale, there is a need to develop a concept for a more effective risk reduction. Examination of the 2002 flood also points to a considerable and urgent need for information on how to provide protection against flooding in an emergency situation, as well as regular information to heighten public awareness about flooding (DKKV publication, 2003).

Clearly, in terms of flood and risk management, the central problem in Elbe is how to provide a sustainable level of flood protection.

3.3.2 Management objectives

As envisaged, the main function of the Elbe prototype DSS is to link selected management measures and scenarios such as dike shifting and land use development to the objectives identified. Therefore, FRA must be carried out to supply appropriate models, tools, strategies and instruments that include flood aspects from disciplinary perspective. Flood management can be categorized into long-term planning versus short-term operation in temporal scale, and large scale (catchment level, the whole Elbe basin in Germany part with length of 500km) versus small scale (section or regional level with length of 50km). Therefore, the management objectives can pertain to four spatial-temporal scales: long-term large spatial scale FRA, long-term small spatial scale, short-term large spatial scale and short-term small spatial scale.

In case of long-term planning, FRA focuses on the change of the future context such as climate change and land use change. These in turn cause changes of the hydrological regime and the economic value at risk area of the river basin. The most immediate and direct approach for such a purpose is the statistical risk analysis (Chapter 2). In case assessment is related with a certain return period of a flood defense system, a hydraulic model should be of help in particular on the analysis of corresponding change of hydraulic characters such as water level ~ discharge relationships. However, it does not necessarily relate to the physically-based approach for FRA. Depending on the indicators that a long-term planning is looking for, as well as the combination of temporal and spatial scales, selection of risk assessment approach is case dependent. For example, if an overview risk distribution is needed for long-term planning, a catchment scale using statistical risk analysis approach is valuable, whereas a long-term local scale using statistical risk assessment is significant for the regional development but may neglect a non-local effect to downstream areas. For such a case, both risk assessment approaches shall be applied.

In the case of short-term management FRA focuses on the assessment of consequence failure of flood defense systems for a certain flood event, which can be combined with the effects of potential flood mitigation measures. FRA serves three management purposes (Kundzewicz and Samuels, 1999):

- *Pre-flood activities* are flood risk management activities related to all causes of flooding. This includes disaster contingency planning to establish evacuation routes, the formulation of critical decision thresholds, improving public service and infrastructure requirements for emergency operations, the construction of flood defense infrastructure (both physical defenses as well as forecasting and warning systems), improved land-use planning , improving public communication and education related to flood risk and the actions to be taken by those involved in case of a flood emergency.

- *Operational flood management* pertains to the detection of the likelihood of a flood (hydro-meteorology), forecasting future river flow conditions from the hydro-meteorological observations, issuing warnings to the appropriate authorities and the public on the nature/severity of the flood, and timing the flood.
- The *post-flood activities* include relief of the immediate needs of those affected by the disaster, the reconstruction of damaged buildings, infrastructure and flood defenses, recovery works to regenerate the environment and the economy in the flooded area, and reassessment of the flood management policy for future improvement.

In addition, integrated water resources management requires flood management to be incorporated into national and international spatial planning and institutional infrastructures.

The objectives and possible environmental and socio-economical indicators in relation to flood management for Elbe River can be summarized as:

- To assess the flood risk under a changing future contexts for long-term planning of the catchment level (including the regional level). The corresponding indicators are: statistical spatial risk distribution that reflects a change in the future context (section 3.2.5); risk level that represents the natural characteristics of the river basin.
- To assess the flood risk associated with scenarios of flooding events. The corresponding indicators are: statistical risk distribution reflecting change of future contexts (long-term planning); for short-term flood mitigation management indicators can be the flood damage in terms of losses of means such as momentary losses, affected people; change of water level and discharge, inundation depth, flooded area.
- To assess the consequence of flood mitigation measures or scenario of flooding event. The corresponding indicators are: the damage reduction in terms of losses of means such as the direct momentary loss, reduction of water level and discharge.

3.3.3 Flood mitigation measures and dike breaks

As the fourth generation of flood management (Green et al., 2000), the present flood management strategy aims for a flood or risk mitigation instead of flood control. Therefore, it is important to identify measures for such a purpose.

The main function of a DSS is to link the *measures* that can be implemented to the objectives. Suggestions for promising measures can be made by the end-users themselves, or the team of researchers designing the model. Flood protection measures incorporated in the Elbe pilot DSS (Matthies et al., 2003) are:

- Dike displacement including dike shifting, dike heightening. The implementation of such a measure can cause a change of hydrological statistics of the flood defense systems such as the return period of a dike, as well as change in geographic conditions of the river.
- Use of retention basins (polders). The function of a retention basin is to provide a temporary storage of a water volume to reduce downstream flood risk. The implementation of retention basins causes a reduction in the water level and discharge at downstream areas.
- Land-use management in the areas at risk.

Depending on the objectives of flood managers and local conditions a dike breach can be considered as a flood event, i.e. a scenario, or as a flood protection measure. In the latter case an intentional dike break can be caused upstream of an economically vulnerable area, thereby sacrificing an area of lower economic importance to lower downstream flood levels. The difference with a retention basin is that there are in general hydraulic structures such as gate to diverge flow into the retention basin, whereas an artificial dike break is an emergency operation. For a large river such as the Yangtze River in China, for example, both retention basins and intentional dike breaks are part of the common flood management procedures. In the Elbe pilot DSS dike breaks at different locations are included as pre-calculated scenarios to show to the DSS users the effects of an uncontrolled dike break for various locations. In this thesis the analysis focuses on the effects of a dike break along the Elbe River as management measure, including the downstream lowering of the water levels.

3.3.4 Scenarios for the future context

Scenarios to describe future context in the terminology of system analysis (Miser and Quade, 1985) are uncertain physical, socio-economic, and other conditions that may affect the system under study, but are beyond control of the decision makers. Examples of scenarios are climate change, population growth, or the Czech input of pollutants. The advantage of using scenarios is that the impacts of certain processes can be accounted for, even if these are not part of the system. This means that the scenarios provide an exogenous input for the model system.

After discussions between modelers and end users from Germany, for FRA, in Elbe_DSS the following scenarios were considered relevant:

- *Socio-economic changes*: represented by the change of land use, socio-economic changes affect all types of socio-economic assessment in flooding.
- *Climate change*: such a change can cause the change of hydrological statistics of the river, which is more related to statistical risk analysis.

3.3.5 Indicators

As mentioned earlier, the cause ~ effect relationships between measures, scenarios and objectives, can be represented through the change of either they hydrological statistics or a change of hydraulic properties (Table 3.1).

Table 3.1 *Cause ~ effect of measures and future contexts*

Contents	Cause	Effect
Measures	Dike displacement	Change of hydrological statistics Change of hydraulic properties
	Retention basin	Change of hydraulic properties
	Dike break	Change of hydraulic properties
Future contexts	Climate change	Change of hydrological statistics
	Land use change	Change of economic properties

Based on the cause ~ effect relationships, the effectiveness of measures can be evaluated together with future changes. In relation to flood management the objectives, indicators categorized as environmental and socio-economical indicators are:

- *Environmental indicators*: inundation depth, flooding area, maximum velocity, hydrograph of the water level and discharge at gauge stations, longitudinal water level profile;
- *Socio-economical indicators*: the annual expected damage (flood risk), damage in terms of losses of means such as monetary losses, number of affected people, risk levels.

It has to be noted that the design of the Elbe_DSS is restricted to four management objective categories: navigability, water quality, flood risk, and ecological quality. For vague reasons different modules should not function independently, but must be integrated. Therefore, formulation of process models is not only depending on the inter-connection of indicators but also constraints from other objectives such as navigation and ecology.

The next step of the qualitative analysis is to identify the relevant processes and variables, as described below.

3.4 Identification of Relevant Processes and Parameters

To obtain a comprehensive understanding of the consequence of flooding - the flood damage, it is important to understand the cause of flooding. Floods can be caused by natural phenomena such as tsunamis, earthquakes, or human disruption such as bombing a dam. However, mostly floods result from the interaction between uncontrollable natural conditions such as climate, weather and sea level, and the characteristics of a river basin such as the topographic, soil and underground water conditions of the river basin, land use patterns, and river morphology. Floods can also occur as the consequence of a failure of flood defense such as overtopping, instability and foundation failure of dams and dikes.

Following information flow in the river, causal relationship of flood damage can be summarized as: rainfall, snowmelt → river discharge → high water level in the river channel → failure of flood defense system → flood damage. This causal relationship is generic and can be used for any flood risk/damage assessment, which also means specification is needed to make it particular serving for different flood management purpose. Thus, the backward searching, as one of the two directions searching method proposed in Chapter 2, should be carried out to identify risk models as well as hydraulic models. Applying the cause ~ effect analysis, the risk models are connected with measures and scenarios using environmental and socio-economic indicators. Based on the results, a conceptual framework of FRA for Elbe_DSS is formulated.

3.4.1 Flood risk assessment approaches

To provide flood risk and damage associated with different purposes, FRA involving both statistical approach and physically based approach, as categorized in Chapter 2, are identified. The statistical risk model provides statistical risk distribution according to the -geometry property and land use distributions, which can be used for long-term planning, whereas the damage model provides damage associated with a flooding event results from different mechanism of flooding, which can be used serve for short-term flood management purpose such as evacuation and flood/risk mitigation, as well as a posterior flood loss estimation – damage assessment after a flooding has occurred (Chapter 2).

Defined as the probability and its consequences, flood risk can be distinguished as two types: statistical risk and flood damage. The effect factors are the exceedance of flood frequency of flood defense systems, and hydraulic factors including inundation and flow velocity associated with a certain flood event. Land use is a key aspect for risk assessment. Thus, requirements from different aspects such as objectives, measures and future contexts, can be represented through the change of hydrological statistics, or the change of physics or both, which can be linked with two types of risk models associated with two types of approaches, namely the statistical approach and the physically-based approach, respectively.

3.4.1.1 Statistical approach

In the statistical approach (e.g. Vose, 1996), flood risk is defined from the combination of the probability of a flood and the consequent damage. Mathematically, it can be expressed as the product of failure probability failure of river defenses, and the estimation of loss damage in case of this failure:

$$R = p_{failure} \times f_{damage} \tag{3.1}$$

where overtopping of a dike is taken as the damage occurrence which is associated with a certain return period of discharge. The expected value of the damage is given by (Stedinger, 1997):

$$< D >_{ij} = \int_{q_{ij}^*}^{\infty} fd(h_{ij}(q)) \times f(q)dq \tag{3.2}$$

Where,

$<D>_{ij}$	Expectation value of percentage flood damage at cell (i,j) [Euro]
q^*_{ij}	Critical discharge, calculated using rating curve at cell (i,j) [m^3/s]:

$$q^*_{ij} = \begin{cases} (\frac{z_{ij}}{a_r})^{1/b_r} & z_{ij} > z_{dike_i} \\ (\frac{z_{dike_i}}{a_r})^{1/b_r} & z_{ij} <= z_{dike_i} \end{cases} \tag{3.3}$$

z_{ij}	Elevation at cell (i,j) [m]
a_r, b_r	Rating curve coefficients at row i, provided by model HEC6 [-]
z_{dikei}	Dike height at row i, applied to both the left-hand and right-hand sides [m]
fd(h(q))	Flood damage function, as a function of inundation depth [%]
$h_{ij}(q)$	Inundation depth in cell (i,j) [m]
f(q)	Probability density function for the Gumbel distribution, used:

$$f(q) = b_g \times e^{-b_g(q-a_g)} \exp(-e^{-b_g(q-a_g)}) \tag{3.4}$$

Using water level ~ discharge relationships, the statistical risk model calculates the expected annual damage at each grid cell. The result of the assessment is an expected value of the percentage flood damage.

Applying the backwards searching following the causal relationships between indicators and processes, the relevant components of in the statistical risk model are as indicated in Figure 3.7.

As shown in Figure 3.7, the backward searching starts from the indicators, in this case two indicators are identified or required, namely the *risk level* aiming to incorporate effects of flow velocity using the risk matrix concept, and the *expected annual damage*, which is the direct result of the statistical risk analysis approach. The use of risk matrix indicates the need for a hydrodynamic model that can produce velocity at the flooded area, where the expected annual damage requires simple water level ~ discharge relationships. This implies a hydraulic model that can at least provide reasonable water level ~ discharge relationships (or water profile along the river). Such model requirement can be satisfied by a steady state hydraulic model. A more complex model for instance a 1D hydrodynamic model will also provide such results, however, comparing the cost in term of model calibration, validation and data demands, a 1D hydrodynamic model is rather expensive. This evaluation is based on the assumption that both models can reach similar accuracy or performance in terms end user requirements. Nevertheless, through the causal relationship as explained above, considering the hydraulic model complexity as the target, the double-direction searching results in the appropriate model complexity.

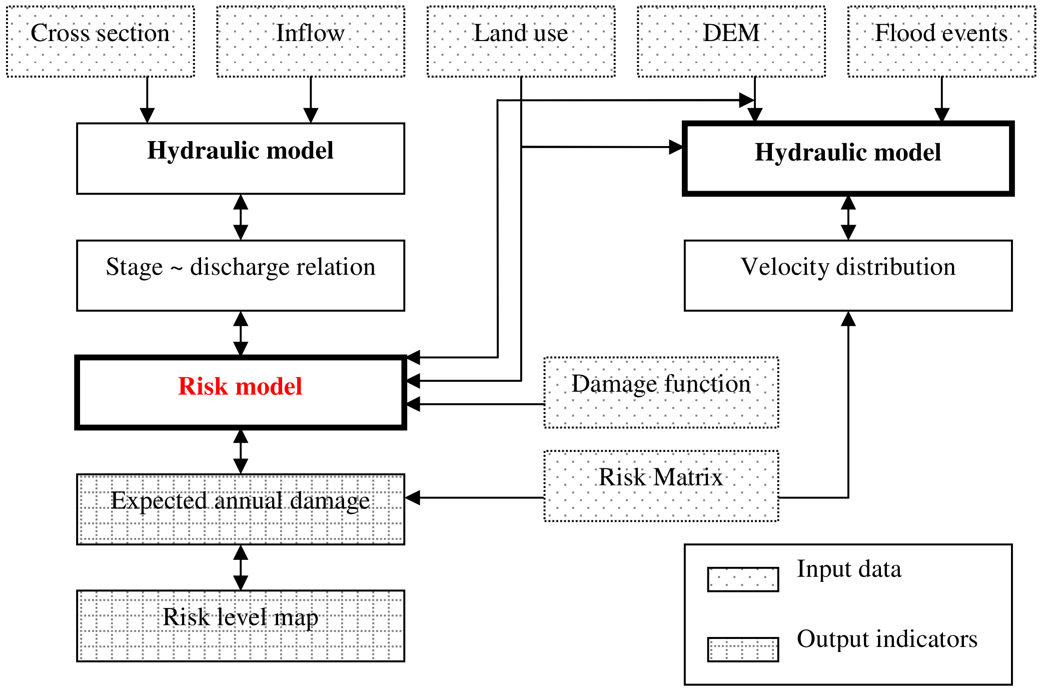

Figure 3.7 *Components of risk model based on the statistical risk-analysis approach*

3.4.1.2 Physically-based approach

The physically-based approach aims to assess flood damage associated with a certain flooding event. The damage model translates inundation depth into monetary loss, which can be used to assess flood damage before and after a flooding has occurred. This model requires computation of the inundation depths associated with a flood event (time series) resulting from the volume of water associated with the occurring time and period of the flooding. A hydrodynamic model is required.

Searching backwards from risk/damage through causal reasoning, the relevant components involved in the damage model are identified as illustrated in Figure 3.8.

The backwards searching in this case is directed by the indicators of flood damage and risk level (if possible), which is by definition flood even based. Therefore, the hydraulic model is more obvious to include a 1D flood routing in the main channel, and at least the continuity conservation law to be applied at the flooded area outside of the river, in which if a detailed velocity distribution is needed a 2D hydrodynamic model is necessary. In general to obtain realistic inundation modeling, a fully 2D hydrodynamic model is needed. However, to obtain a rapid assessment for inundation depth without considering flow velocity effects, an approximation can be made using GIS technology. Nevertheless, the backwards searching meets the forward searching at a hydrodynamic model which indicates the appropriate complexity of the hydraulic model for the physically-based risk assessment approach.

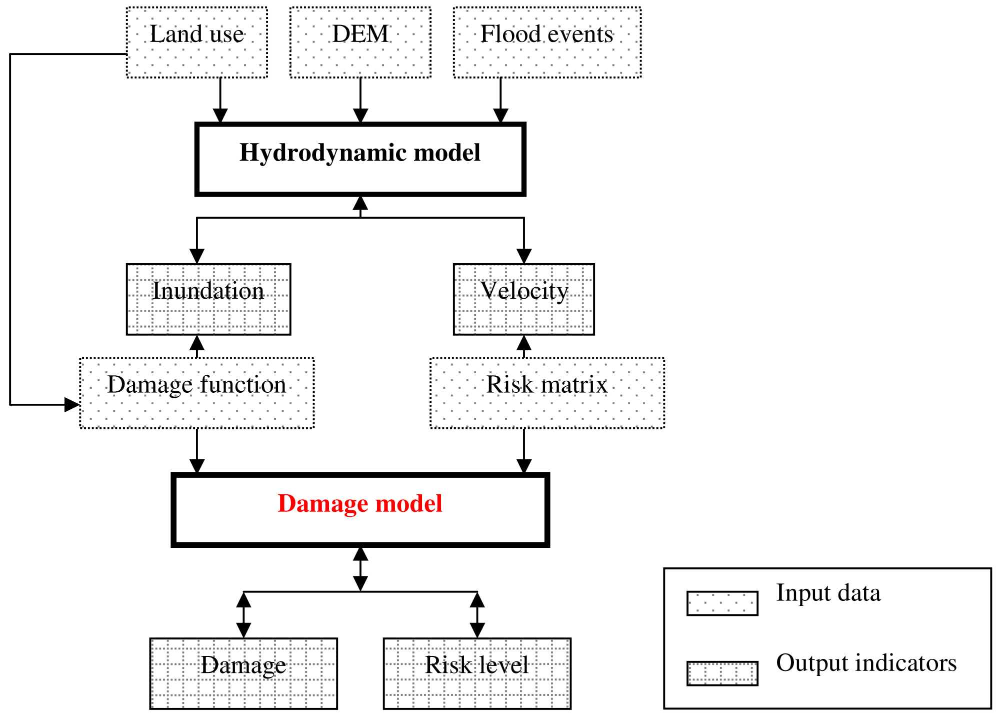

Figure 3.8 *Components of damage model based on physically-based approach*

An important input of a physically-based approach is the flood event at upstream boundary. For example, to study the impact of different flood management measures such as an intentional dike break aiming for risk reduction, generating of an extreme flood event at the upstream boundary is essential. The 35 years hydrological time series at Elbe, however, the high-quality representative hydrological data pertain to a short time series (Helms et al., 2002a, b). Therefore, an artificially generated flood event is needed.

In general three variables characterize an artificial flood event: the peak value, the duration of the flood event, and the shape of the flood wave. They can be determined following three major steps as described below.

<u>Generate peak discharge</u>
This can be made using statistical distribution of the annual maximum discharge. In general, a Gumbel distribution is applied. The peak discharge is then calculated as (Shaw, 1994):

$$Q_{pT} = a_g - \frac{1}{b_g} \ln\left(\ln\left[\frac{T}{T-1} \right] \right) \qquad (3.5)$$

where,
Q_{PT} Peak discharge [m^3/s] associated with return period T [year]
a_g, b_g Gumbel distribution coefficients fitted using the historical data [-]

Determine flood duration

The duration of a flood event is determined from a historic time series. A functional expression between the peak discharge and its duration in days is generated follows steps of:

- In view of the uneven contribution of the flow, the discharge is stratified into 100 m^3/s intervals between zero discharge and the maximum value, as it observed in the historic data. The duration D_i assigned to each discharge value Q_i is the average duration of the discharge values in each interval:

$$D_i = \frac{1}{m}\sum_{j=1}^{m} Dq_j \qquad where \quad Q_{i-1} \le q < Q_i \tag{3.6}$$

  where D_i is the new duration associated with each ranged Q_i, Dq_j is the original flood duration from historical data.
- These set of ranged duration is fitted using a power function, therefore discharge values expressed as:

$$Q_p = a_p D^{b_p} \tag{3.7}$$

  Where Q_p is the peak value of discharge in m^3/s, D is the flood duration in days, a_p and b_p are the fitted power function coefficients.

Determining the shape of the flood event

This can be made using a simple scaling method. In this method, flood events are detected from the historical data, and are normalized to fit the range [0, 1]. The final shape curve is obtained from the smoothed mean value of the normalized wave shapes.

3.4.2 Selection of hydraulic models

As shown in Figure 3.7 and 3.8, the essential aspect of the risk model is the choice of the most appropriate hydraulic models.

The statistical risk model uses a stage ~ discharge relationships as output of hydraulic model. Therefore, for a river channel simulation a simple one dimensional steady state hydraulic model would satisfy such a requirement, which can simulate water profiles along the river. Interpolation of the rating curves generated with historical data is not recommended, because of inconsistency of the depth ~ damage relationships caused by the future physical changes, which cannot be represented by the historical rating curve. Therefore, a physically-based hydraulic model is required to represent such a change.

In the physically-based approach, due to the flow dynamics with more than one velocity components in the floodplain area, as well as damage indicators requirements in terms of the reduction of water level and discharge at a downstream area, a two-dimensional hydrodynamic model should be used.

A question might arise: why not use a 2D model which will satisfy all the requirements of both models? The answer is that it is possible theoretically. However, as FRA are usually carried out to aid decision making for the implementation of flood mitigation measures, timing is one of the key criterion that a FRA should fulfill. Thus, it might be not suitable from practical point of view to directly run the 2D model in real time when a rapid assessment is needed, because for example the experience shows that for a 200 year return period flood (covers 20 days flood duration) with time step of 15 minutes, for a area of 10km×5km, cell size of 50 meters, the computation time is about 9hours with normal PC configuration. A rapid assessment can be obtained with interpolation of a set of pre-calculated scenarios. Drawback is that the combinations of scenarios are limited, and it is lack of flexibility, apart from the uncertainty contributed from interpolation procedure. Therefore, it is not very practical to run directly a 2D model for rapid risk assessment which requires a short time (for example 10 minutes) response.

To avoid the large computational load particularly time demand, a quasi-2D approach, i.e. an alternative is to have simplified floodplain inundation modeling which does not require computation of momentum at floodplain areas, is needed. An example of such a quasi-2D approach is LISFLOOD (Bates and De Roo, 2000; Horritt and Bates, 2002). Quasi-2D can be also an approximation using GIS technology.

3.4.3 Risk matrix

As discussed previously, the flow velocity plays a vital role for damage caused by flooding. However quantitative relationships between flow velocity and damage/risk are missing. To cope with this problem, various forms of risk mapping using risk matrix have been applied (Du Plessis, 2000; Adriaans, 2001; Stephenson, 2002; Roos, 2003; Fattorelli et al., 2003; Vrouwenvelder et al., 2003; Huang et al, 2004). Regardless what combination the velocity is combined with, i.e., with probability of occurrence, or with the inundation depth, or other risk indicators such as potential damage expressed as monetary loss, these methods all require a classification of the involved risk parameter. Criteria of classification are therefore needed.

For example following the Adige River Authority's requirements a criterion to identify flood risk based on hydraulic factors, Fattorelli et al. (2003) classified the risk level into four classes combing two hydraulic factors, the inundation depth and maximum flow velocity. In this study, instead of using flood hazard defined as the sets of combinations of the inundation depth and maximum flow velocity subject to different return period, the percentage damage caused by inundation associated with land use, is used as the verse variable to velocity. Thus, the risk matrix is obtained by combining the classified annual expected damage and normalized flow velocity.

Four qualitative risk levels are distinguished, each reflects the necessity of flood mitigation activities depending on the classified damage and flow velocity (Table 3.2).

Table 3.2 *Risk matrix*

Velocity index (%) / Expected annual damage (Euros)	(0,25]	(25, 50]	(50,75]	(75,100]
0-1000	R1	R2	R3	R4
1000-10,000	R2	R2	R3	R4
10,000-100,000	R3	R3	R3	R4
>100,000	R4	R4	R4	R4

Note: the classification of risk matrix is case-dependent and requires agreement with end users to correspond flood management activities.

As indicated in Table 3.2, the spatial distribution of flow velocity is an indication of momentum characteristics. It has been found difficult to obtain the probability distribution of flow velocity which is case-dependent, i.e. depends on the combination of probability of flood in the river and the probability of failure of flood defense, the physical process of the failure (i.e. either through dike break or overtopping). To obtain a general representation of spatial distribution of velocity, a normalized maximum velocity obtained from a number of 2D hydrodynamic simulations associated with failure of flood defense system such as a dike break. For each cell *ij*, the velocity index VI_{ij} is calculated using the expression of:

$$VI_{ij} = \frac{1}{m_e} \sum_{k=1}^{m_e} v_{ij,k}^{N} \text{ , } m_e = 1,2,\ldots,ne \tag{3.8}$$

where m_e is the number of effective flood events which have velocity values at cell *ij*, *ne* is the number of total amount of flood event, $v_{ij,k}^{N}$ is the normalized velocity at each cell *ij* for flood event *k*, calculated as:

$$v_{ij}^{N} = \frac{v_{ij}}{v_{\max}^{s}} \tag{3.9}$$

Where, v_{ij} [m/s] is flow velocity at cell *ij*, $v_{\max}^{s}$ [m/s] is the spatial maximum flow velocity.

Hence, a velocity index is calculated. Main uncertainty of the representativeness of this method is the number of flood events, which formulates the sample base of such a statistical approach. Simplification might be needed due to the large 2D hydrodynamic computation that is required for a realistic inundation modeling.

The risk matrix accounts for the additional damage caused by velocity. It is of more operational sense, and in turn more for flooding warning purpose in compared to the long-term planning.

3.4.4 Conceptual framework for IFRA for Elbe_DSS

To distinguish from models, objectives, future context and measures can be regarded as external system components. Following the causal relationships between risk models and hydraulic models incorporating measures and future context, the risk models developed based on two approaches of the statistical risk-based approach and the physically-based approach, are combined with external system components.

However, such a framework consisting of disciplinary process such as hydraulic and hydrology can only represent the instantaneous impact of the implementation of measures and the change of future contexts. To achieve flood management objectives, the distinction should be made between combination of measures and future contexts. As explained in the literature, UA can be applied to serve for such an impact assessment (De Blois and Wind, 1996; Al-Futaisi and Stedinger, 1999; NRC, 2000; De Roo et al., 2003; De Kok and Wind, 2003). Therefore, UA should be applied to distinguish consequences of changes.

Based on the above analysis, a framework of conceptual FRA for Elbe_DSS is formulated with two risk models based on the statistical approach and the physically-based approach, respectively, as shown in Figure 3.9.

In the framework, hydraulic models includes a simple steady state hydraulic model that can provide water level ~ discharge relationships for the statistical risk computation, and a 2D hydrodynamic model which can simulate flood events and provides inundation and velocity map at the floodplain areas for the physically-based approach. The selection of a 2D model is based on the requirement of velocity at the floodplain area. In case velocity is not needed, a storage-approximation (quasi-2D) at floodplain area can be an alternative for a rapid inundation depth computation.

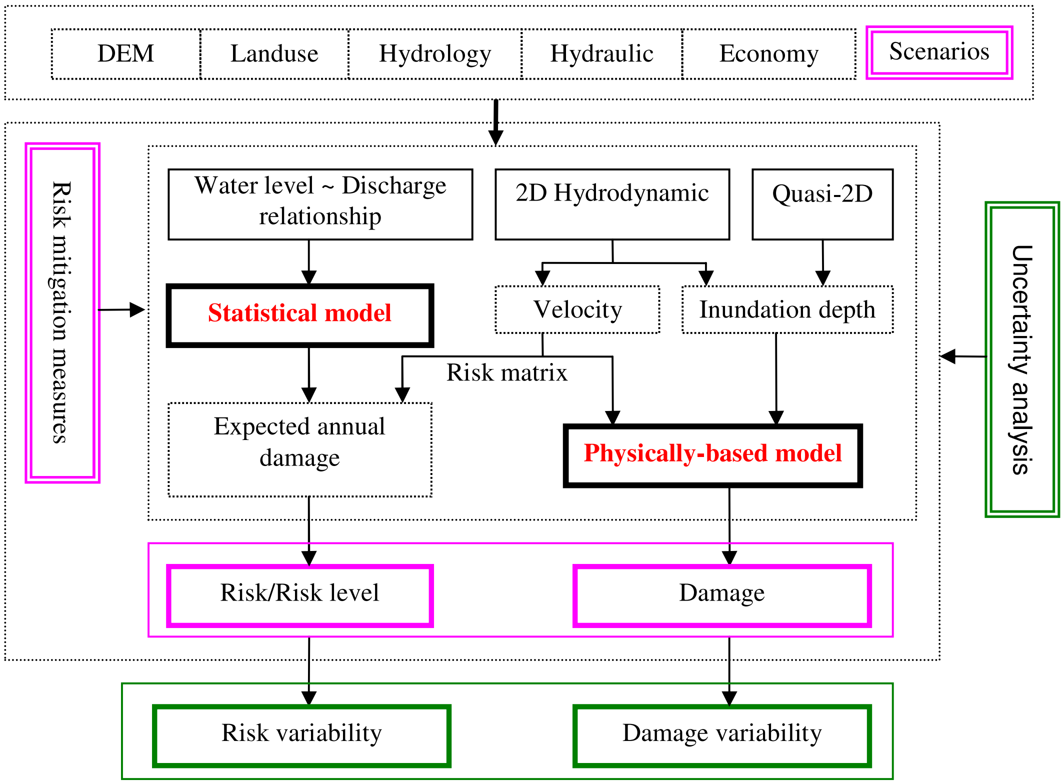

Figure 3.9 *Conceptual IFRA system for Elbe_DSS*

Measures are applied externally to the risk models through the hydraulic models. Scenarios (future context) are treated as the change of input data which can be simulated by hydraulic models and risk models. UA is a post-evaluation process which provides consequence/difference of implementation of measures or the change of future contexts.

Data demand can be obtained following the information flow as indicated in the framework. The IFRA starts from hydraulic level, which uses hydrological data such as discharge and water level time series as boundary conditions. Other necessary data for IFRA are: a DEM, land use data, and data of flood defense system such as dike data including locations and height.

3.5 Conclusions

The Elbe River system has been introduced, including a description of physiography of the region, the flood risk problem, main management objectives, and potential measures that the authorities could use to reduce flood risk along the river. A distinction is made between short- and long-term planning. In addition a risk assessment can be aimed at predicting the flood risk at the local or non-local scale.

The double-direction search and qualitative system analysis of Chapter 2 were combined into a conceptual framework for IFRA, using flood risk along the Elbe river as case study. The IFRA framework can be used to decide which type of risk assessment approach, statistical or physically-based, and which type of hydraulic model are most appropriate to assess flood risk. For a statistical risk assessment the computational load makes application of 2D hydrodynamic models less desirable, and 1D or quasi-2D models are to be preferred. For a physically-based approach the choice between a fully 2D or a less complex hydrodynamic model should depend on the purpose of the assessment, local- vs. non-local analysis, short- versus long-term analysis, and in particular whether information on flow velocities is needed.

The risk matrix concept benefits the framework of IFRA by adding effects caused by inundation flow velocities. The principle of a risk matrix can be used in both a statistical approach and a physically-based approach depending on the different flood management objectives and activities.

Chapter 4

Application of IFRA Framework at the Local Scale

The purpose of this chapter is to test the applicability of the proposed IFRA framework for FRA at the local scale. The region near the town of Sandau (Elbe km 412-422) serves as case study, and is briefly introduced in the beginning of the chapter. Land use in the area is largely agricultural and forest, but the two towns of Sandau and Havelberg are potentially vulnerable to flood risk. As has been recommended in Chapter 3 a steady state hydraulic model is the best choice for a conventional statistical approach, whereas a hydrodynamic model is better for the physically-based risk assessment. Here the two approaches will be compared.

For the *statistical* approach (section 3.4.1.1) two different indicators for presenting flood risk to decision makers are compared. First, the expected annual flood damage (in euro) resulting from the maximum inundation depths reached during a flood. A drawback of this indicator is that flow velocity cannot be incorporated, because of the lack of corresponding damage functions. The second indicator assesses the risk level qualitatively based on a risk matrix, combining the effect of flood damage and flow velocity. Using the risk matrix, a different spatial pattern for the flood risk around Sandau is obtained.

The risk matrix, however, is considered to be less useful for the *physically-based* approach (section 3.4.1.2), in view of the short-term planning purpose focusing on demonstration of the effects of a particular flood event. Application of the physically-based approach require an artificial 200-year flood event for the upstream gauge station at Tangermünde (Elbe km 388) generated using the simple scaling method proposed in Chapter 3. The physically-based approach results in an inundated damage map for the chosen flood event, and a map of maximum flow velocity.

Finally, in order to evaluate how significant the difference between the impact of measures, *Uncertainty Analyses (UA)* are carried out using the two risk assessment approaches. For the statistical approach the significance of the effect of dike presence is examined. For the physically-based approach two scenarios are compared: the situation with and without an artificial dike break.

4.1 Introduction

The conceptual framework presented in Chapter 3 is facilitated with selected hydraulic models and data, as shown in Figure 4.1.

As shown in Figure 4.1, two risk assessment approaches are used, namely the statistical approach, and the physically-based approach. The risk-analysis based model provides the expected value of the annual flood damage, which can be carried out for a basin scale such as the whole Elbe River basin, whereas the physically-based approach is used to assess the flood damage for a specific flood event, and more suitable, in terms of computational loads and timing, for smaller scale such as 50 km of a river reach. Both risk models are integrated with the hydraulic models selected during the qualitative study, namely the steady state model HEC6, and the floodplain hydrodynamic model SOBEK1D2D, for the statistical and physically-based approach respectively. The selection of these two hydraulic models is described in section 4.3.

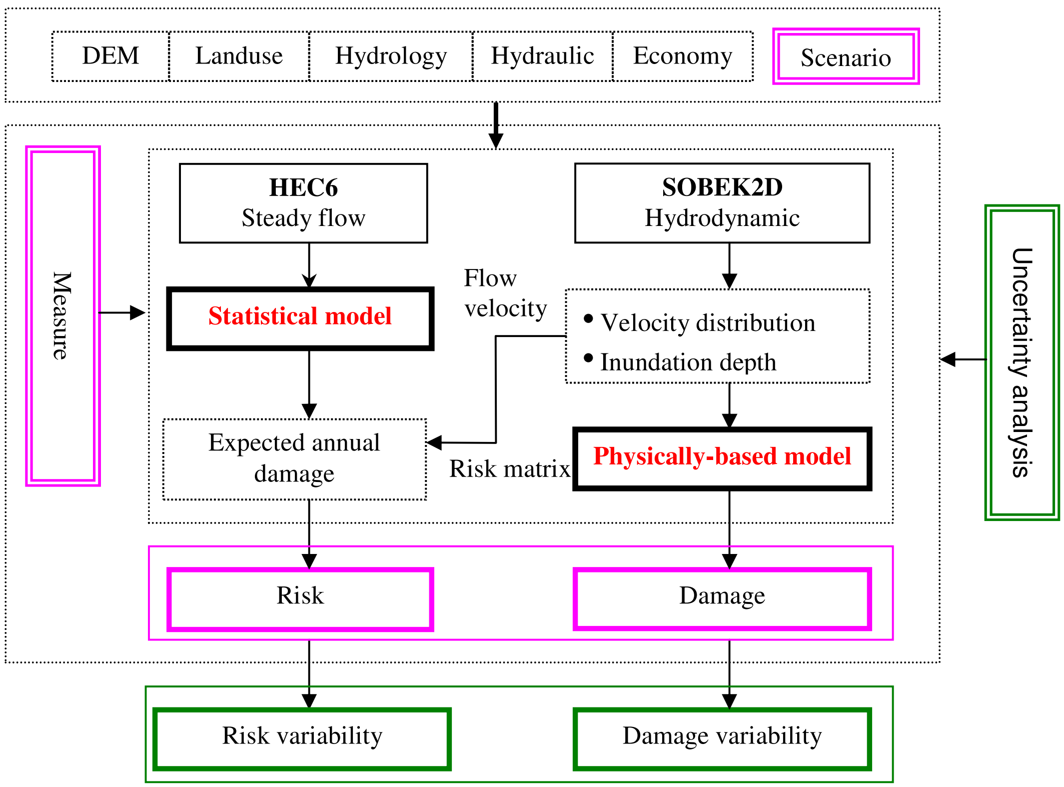

Figure 4.1 *Outline of flood damage assessment system*

Because of the data availability, the area near the town of Sandau in Germany at the Elbe River has been selected as the case study area. To assess the effectiveness of dike presence, four scenarios pertaining to two measures are set up using the IFRA. These are: 1) measure of removing dikes in the study area; and 2) an intentional artificial dike break. Both the statistical and physically-based approaches are applied. MCS is applied to propagate uncertainties through the system for each of the scenarios, and for each FRA approach. The following sections introduce the case study area and present the data condition for the modeling area of Sandau along the Elbe River (Germany).

4.2 Data and Model Conditions

4.2.1 The Sandau area

As IFRA has a broad requirement with respect to the data including DEM, land use, hydrology and hydraulic data, the selection of modeling area was mainly based on the data availability in the development of IFRA for Elbe_DSS (Matthies, et al., 2003). The region of Sandau near

Tangermünde (Elbe km 388), has been chosen (as the case study) for a quantitative analysis of the IFRA.

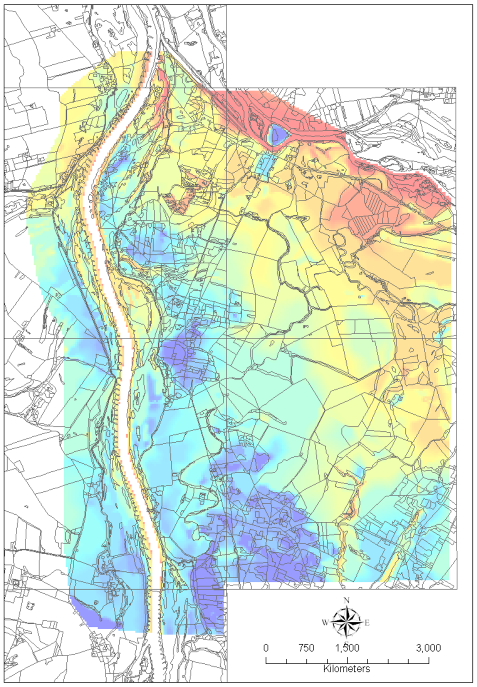

Figure 4.2 *Modeling area, Sandau*

Figure 4.2 shows details of the topography of the Sandau area. In the shadowed area is the available DEM data (Otte-Witte et al., 2002). In the area, the elevation ranges between 22.65 m and 38.67 m (NH) with a mean value of 26.73 m and spatial standard deviation of 1.51 m.

The land use data are based on the European CORINE database (EEA, 2002), which includes land use data from fifteen EU Member States as well as some other European and North African countries. It is at an original scale of 1: 100 000 and using 44 classes of the 3-level CORINE nomenclature (Appendix I).

The proportional distribution of land use in the study area is shown in Figure 4.3, illustrating that Sandau is a typical rural area. Most of the area is occupied with agriculture (48%) and forest (26%). Only a very small area (5%) is urbanized and has a higher economic value.

As described in Chapter 3, because damage functions for the 44 land use classes are not available, the CORINE classes have been reclassified into seven risk relevant types of land use with associated flood damage functions based on their economic similarities. A similar reclassification has been made for the hydraulic roughness (Appendix II).

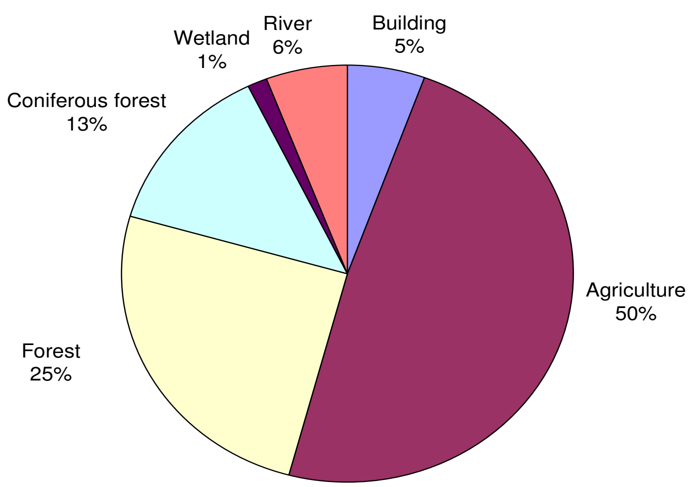

Figure 4.3 *Land use distribution for Sandau area, based on the CORINE land use classification*

4.2.2 Dike data

In the area of Sandau due to the lack of actual dike data, the design dike height between Tangermünde and Wittenberge (km388.2-456.2) is used in the computation. The design dike height corresponds to a return period of 10 years for overtopping on the left-hand side and 100 years for the right-hand side (IKSE, 2001) (Chapter3). It is calculated as the design water level H_d corresponding to a return period with an additional height of freeboard H_f (Figure 4.4).

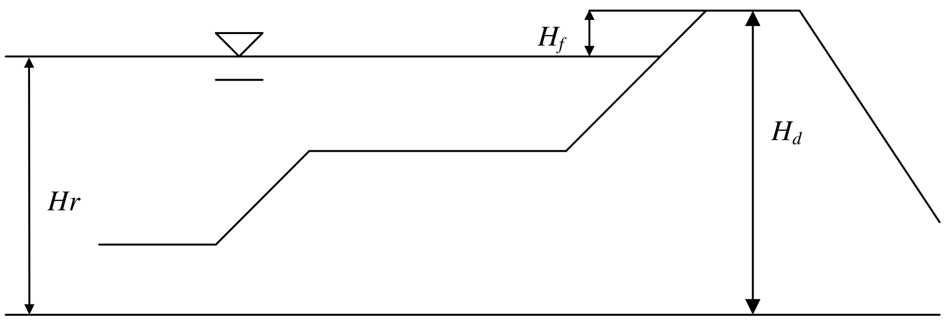

Figure 4.4 *Design dike height computation scheme (IKSE, 2001). H_f is the freeboard, Hr is water level associates with food return period T*

The dike height parameters consist of geographic information such as the location and coordinates assigned to each river km. The calculation of the dike height at different location follows the same scheme as shown in Figure 4.4. The dike height is expressed as:

$$H_d = H_f + H_r \tag{4.1}$$

where,

H_d Dike height [m]

H_f Freeboard [m], calculated as $H_f = H_{wave} + H_{wind} + H_{additional}$ (4.2)

Hr Water level [m], associates with flood return period T, calculated using rating curve:

$$Hr = f(T) \tag{4.3}$$

The designed dike height is adopted in the risk assessment. Because the dike height is smaller at the left-hand side reflected by a smaller design return period for overtopping in years, it may affect uncertainly analysis this dominating effect can hide the contributions of other uncertainties such as hydraulic model structure (numerical schemes), hydraulic parameters such as roughness, or DEM data.

4.2.3 Artificial flood event

An extreme flood event at the upstream boundary is required for the simulation of a dike break. However, in the modeling area, at the gauge station of Tangermünde, the available year discharge time series for the period of 1960-1995, has the maximum discharge of 3259 m^3/s corresponding to a return period of 28 years, according to the Gumbel distribution. For more extreme flood events an artificially generated flood event is needed.

Following the steps proposed in Chapter 3.4.1.2, to generate a flood event of 200-year return period, the peak value is estimated as 4450m^3/s, using the Gumbel distribution (Eq. 3.5). The duration ~ discharge curve is obtained through fitting the power function, expressed as:

$$Q = 9.643 D^{1.66} \tag{4.4}$$

Where Q is the peak value of discharge in m^3/s, D is the flood duration in days.

Using the scaling method introduced in Chapter 3.4.1.2, together with the peak value and duration estimated using Eq. 4.4, a 200-year flood event has been obtained for the gauge station Tangermünde (Figure 4.5).

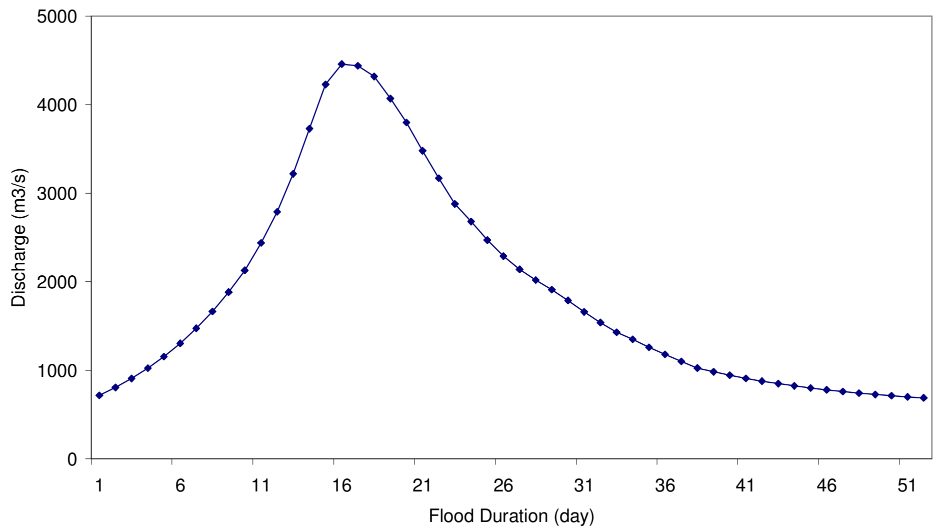

Figure 4.5 *Artificial time series of daily average discharge for a 200-year return period at Tangermünde gauge station (Elbe km388.2)*

It is understandable that the uncertainties are introduced in every step of the procedure, due to the use of the Gumbel distribution, the ranging method used in the determination of flood duration, and the simple scaling method used to define the flood wave shape. These uncertainties are resulted from the hydrological boundary conditions.

4.2.4 Flood damage functions

The core of IFRA is a functional relationship between the inundation depth and the flood damage (White, 1945), expressed as a percentage of the maximum monetary loss – or the potential damage - for each land use. Each land use is categorized according to standards used in the European CORINE land cover data (European Environment Agency, 2002).

Due to the difficulties in the estimation of flood losses, it is difficult to obtain accurate flood damage functions. Flood loss estimates require substantial research resources, which are rarely available. Moreover, losses are usually reported in different ways, the difference between direct and indirect damage further complicates the problem, which makes the flood damage functions difficult to interpret.

In this thesis, the depth ~ damage curves are obtained from different sources (Kok, 2001; IKSR, 2001; Van der Sande, 2001) (Appendix III). Some functions pertain to historical flooding events, such as the 1953 coastal flood in the Dutch Delta region. Other functions have been developed to assess the potential flood damage for the River Meuse (Kok, 2001) or Rhine (IKSR, 2001). Figure 4.6 shows the depth ~ damage curves for five land use classes: urban area, industry, traffic, agriculture and forest, which are reclassified from CORINE data based on their economic similarity.

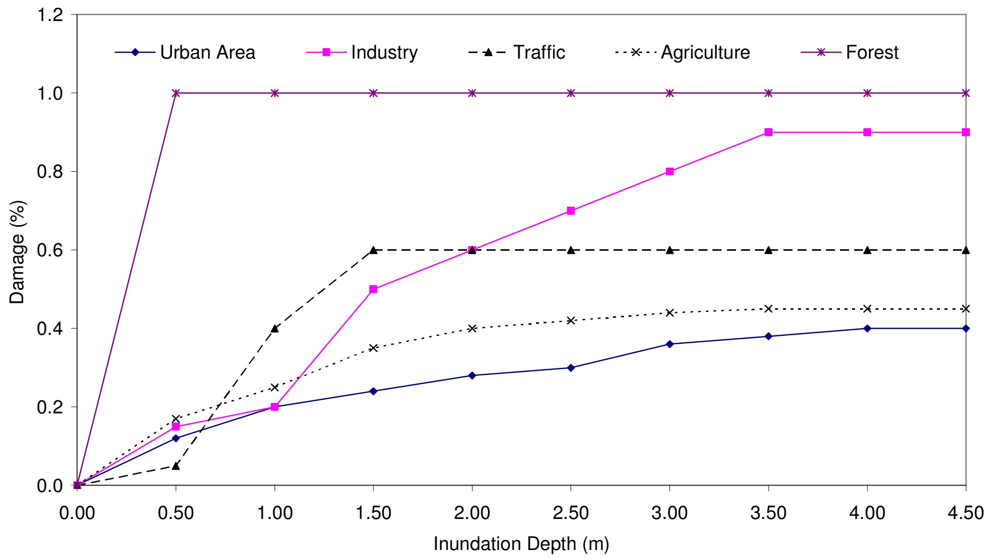

Figure 4.6 *Depth ~ damage curve for urban area, industry, traffic, agriculture and forest (Kok, 2001; IKSR, 2001; Van der Sande, 2001)*

The potential damage or the maximum monetary damage of each land use is location dependent. It is also subject to economic changes such as a change of the inflation rate. The maximum monetary loss to each type of land use caused by floods is listed in Table 4.1. The results are measured in the unit of Euro/m^2 at the German 2001 price level (IKSR, 2001).

Table 4.1 *Potential damage in Euro/m^2 at the price level of 2001 in Germany (IKSR, 2001)*

Land use	Potential damage (Euro/m^2)
Urban area	329.53
Industry	252.48
Traffic,	238.12
Agriculture	12.93
Forest	1.14

4.2.5 GIS data

Several GIS data are used in the inundation modeling, namely DEM, land use data, and roughness map. They were collected and standardized according to the project requirement such as projection method and cell resolution. Those data are exported from GIS format into ASCII raster data, and are used in the models involved in the IFRA.

DEM: To obtain inter-consistency of data, pre-processing was necessary. The original 10m-resolution DEM (Otte-Witte et al., 2002) has captured the geometry characteristics of the study area well, however for this spatial resolution, the hydrodynamic simulation is time consuming. To reduce the computing time and at the same time to keep the main characteristics of the modeling area, a 50m-resolution is finally considered to be appropriate, as for 10km long area more than 100m has been found too coarse to describe the hydraulic and hydrological characteristics. A mean value of neighbor cells is used for the aggregation of the DEM.

Land use: The land use data are based on the CORINE (European Environment Agency, 2002). It provides land cover database for the fifteen EU Member States as well as European and North African countries, at an original scale of 1: 100 000, using 44 classes of the 3-level CORINE nomenclature (Appendix I). Due to the lack of detailed land use ~ flood damage functions and land use ~ roughness relationships, the 44 classes land use have been reclassified into seven types of risk-relevant land use classes to be used in the flood damage functions (Appendix II).

Roughness: Hydraulic roughness values are collected according to resistance similarities of land use types listed in literatures (Chow, 1959; Beasley and Huggins 1982; De Roo, 1999) (Appendix II). For UA of the 2D hydrodynamic model, to save computation time, reclassification of land use has also been made for the hydraulic roughness values based on the resistance similarities (Appendix II).

4.2.6 Hydraulic models

In the study, two models, the HEC 6 steady-flow hydraulic model (HEC6 User's Manual, 1993), and the SOBEK1D2D hydrodynamic model (Verwey, 2001; Stelling et al., 1998; Stelling and Duinmeijer, 2003) are selected.

4.2.6.1 HEC6

As a steady-state hydraulic model, HEC6 provides stage ~ discharge relationships that are used for the statistical risk assessment approach. Hec-6 is a one-dimensional movable boundary open channel flow numerical model that was developed at the Corps of Engineers Tulsa District office (HEC6 User's Manual, 1993). The model is designed to simulate long-term trends of scour and/or deposition in a stream channel that might result from modifying the frequency and duration of the water discharge and/or stage, or from modifying the channel geometry. Due to morphological processes, the bed level changes in time. In this research a constant value for the bed level (fixed bed calculation in HEC6) is used.

The computational procedure is based on the solution of the one-dimensional energy equation with energy loss due to friction evaluated with Manning's equation. The unknown water surface elevation at a cross section is determined by an *iterative* solution. Technical details of HEC6 can be found in Appendix IV.

HEC6 has been calibrated previously for the complete Elbe River in Germany (Otte-Witte et al, 2002). The results can be used immediately, which makes the choice obvious.

The necessary input data for the model includes: cross-section geometry at representative locations, discharge for an upstream starting point, water level data for representative locations (for calibration), rating curve for the upstream cross-section (optional); and roughness coefficients. Model output consists of the water level at all locations of interest.

4.2.6.2 SOBEK1D2D

In contrast to a steady state model, SOBEK1D2D is a hydrodynamic floodplain module of SOBEK Rural package developed in WL|Delft hydraulics, Delft, the Netherlands. The model solves the two-dimensional shallow water equations using the so-called "Delft Scheme" (Stelling et al., 1998; Stelling and Duinmeijer, 2003). Details of the SOBEK1D2D model equations and Delft scheme can be found in Appendix V.

This model has been applied for inundation modeling and has been found robust and effective (e.g. Hesselink et al, 2003; Fattorelli et al., 2003). The readily availability of the model and existing experience with SOBEK1D2D, also support the selection of SOBEK1D2D. As another alternative, the 2D hydraulic model WAQUA has been considered during the feasibility study preceding the design of the DSS for the Elbe (De Kok, et al., 2001). However, this model requires a Unix operation system, which is incompatible with other models developed for operation in a Windows environment.

The main model inputs for the overland module of SOBEK1D2D are: a DEM of the main channel and floodplains, water level (or discharge) time series at the model boundaries and representative locations at the modeling area, and roughness data associated with land use type. The overland module provides maps of the maximum water level and water depth, and maximum flow velocity in the inundated area.

4.3 Flood Risk Assessment Results

With the risk models developed based on the statistical approach and the physically-based approach (Chapter 3), using hydraulic model HEC6 and SOBEK1D2D correspondingly, FRA at Sandua area have been carried out.

4.3.1 Statistical approach

Two scenarios are set up for the statistical risk model at the modeling area of Sandau: (a) the case with the presence of the dike, and (b) the case without dike protection. Using the statistical approach described in Chapter 3, two risk maps of the expected annual damage are obtained for these two scenarios (Figure 4.7a-b).

Figure 4.7 presents expected annual damage of the area at risk. As shown in the risk maps, when there is a dike, the risk at the left-hand side of the river is significantly higher than at the right-hand side (Figure 4.7a). This is mainly due to the lower design level of the left dike compared to that of the right dike. A risk map of the natural situation without a dike is obtained by removing the dike at both sides (Figure 4.7b). It shows the quantitative impact of dike protection, that is, without dike protection the risk is much higher.

Figure 4.7b also indicates the spatial distribution of risk depends on elevation and land use, which is the original dike risk distribution before the dike was constructed. This type of risk map can be used to assess qualitatively the risk reduction resulting from building a dike, or other flood defense system. Two towns of Sandau and Havelberg are also indicated by the dark area near the central and top-right corner of the risk map.

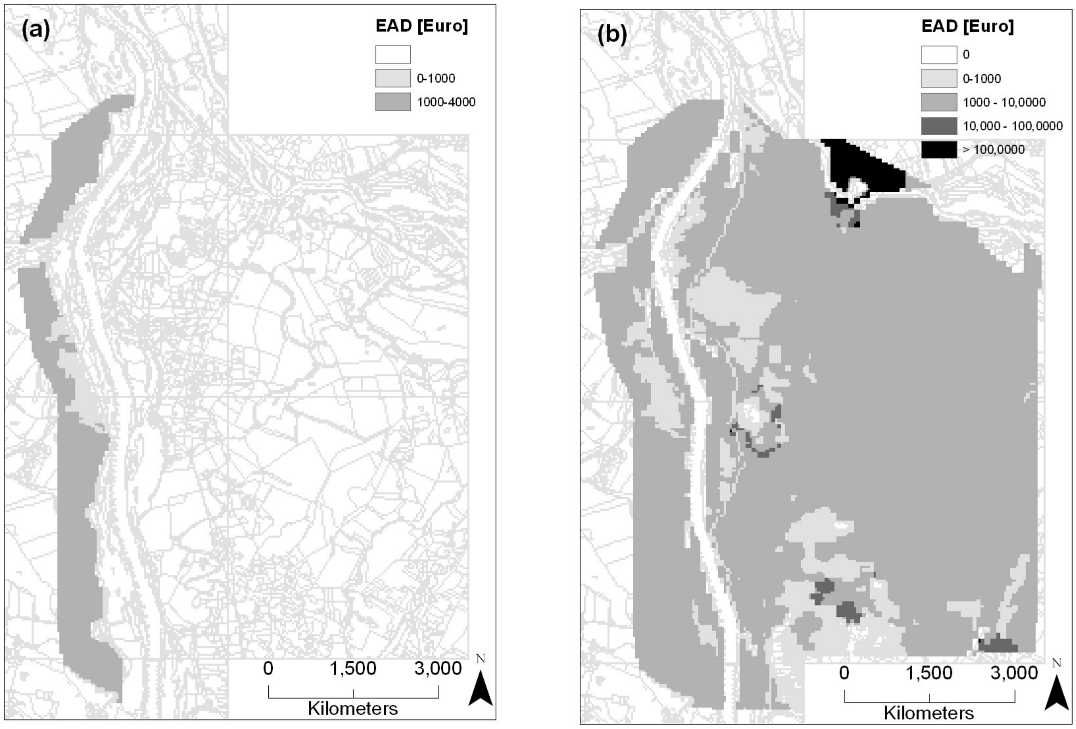

Figure 4.7 *Expected Annual Damage (EAD) in Euro per cell for: (a) with dike protection and (b) without dike. The flood risk is classified into four levels: 0-1000 euros, 1000-10,000 euros, 10,000-100,000, and > 100,000 euros. The maximum risk is $3.85*10^3$ euros and $1.67*10^5$ euros, for case (a) and (b), respectively*

To improve these risk maps by incorporating the effect of more hydraulic factors, a risk level map is made, based on the combination of the expected annual damage and the normalized maximum velocity using the risk matrix presented in Chapter 3, Table 3.2.

To apply the risk matrix, a spatial velocity distribution is important. However it is difficult to determine such a velocity map due to the reasons of: 1) the probability distribution of velocity is dependent on the combination of discharge probability and probability of overtopping of dikes, it is difficult to determine probability distribution of velocity using statistical approach; 2) without hydrodynamic computation it is difficult to obtain a realistic spatial velocity distribution based on GIS and hydrological data only. Thus, a simplification has been made to obtain the velocity distribution using different flood events simulated using SOBEK1D2D. Due to the large computations required for the 2D simulation, only three flood events are taken into account, namely Q28, Q50 and Q200, with return periods of 28 years, 50 years and 200 years.

To represent a normalized spatial distribution of the maximum flow velocity in the flooded area, the normalized average of the maximum flow velocity is used for each event. A weighted average value is then calculated as the indication of velocity in each cell (50 x 50 m).

Finally, two risk maps are obtained: the risk level map of the expected annual damage, which can be regarded as risk level with zero velocity effect (the first column of the risk matrix as shown in Table 3.2) (Figure 4.8.a), and the risk level map based on the matrix of Table 3.2 (Figure 4.8b).

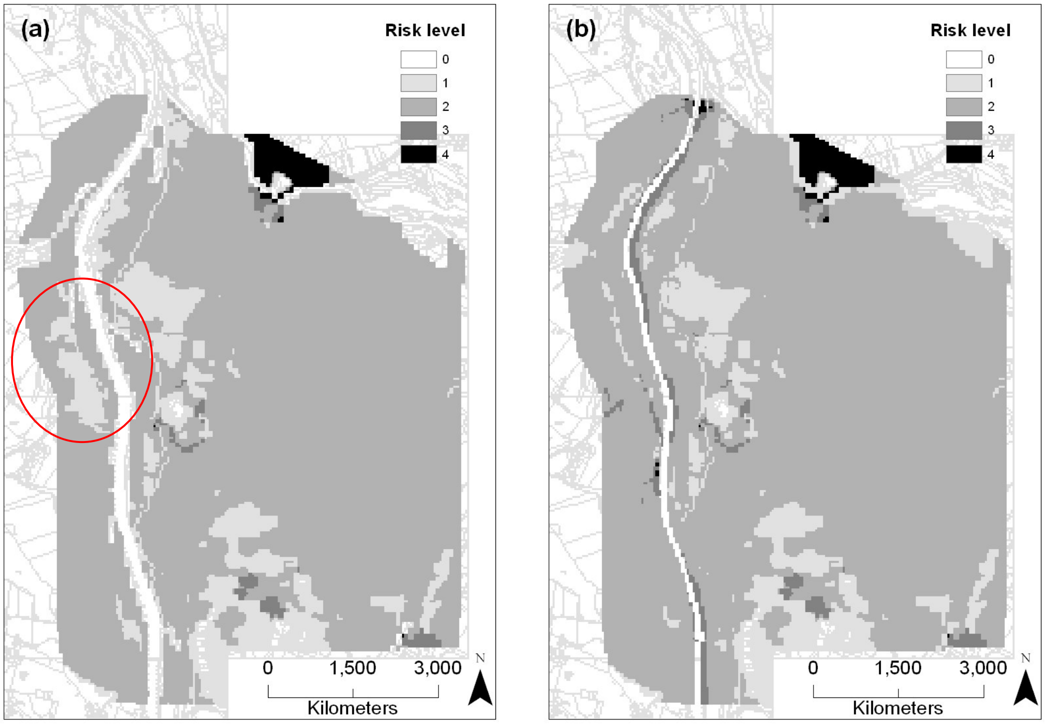

Figure 4.8 *Risk level maps of statistical approach using the risk matrix (Table 3.2): (a) classification of expected annual damage (Euros) without velocity effect; and (b) combining expected annual damage with flow velocity*

As indicated in Figure 4.8a-b, the application of a risk matrix has changed the risk distribution on the left side bank. Differences found are: 1) risk level increased from R1 to R2 in the area indicated by the circle; and 2) risk level increased from R2 to R3 at the right hand side directly next to the river. This demonstrates the additional effect of flow velocity.

Other differences can also be found in the percentage contribution of the risk levels to the overall risk, for each risk level it is calculated from the summed over all cells over all cells with a certain risk level (Figure 4.9). For example, the percentage of risk level R3 increases from 6.13% to 64.44%, while for risk level R1 it decreases from 27.70% to 0.50%.

The results show that by including the velocity effect the risk level distribution changes. In addition, the summed total risk is increased. Thus, with the risk level map, a clearer attention can be paid to the need for the implementation of a flood defense system, such as a retention basin. This could also improve people's awareness of risk by pointing out some high risk area due to flood velocity which might not appear on a risk map considering inundation depths only.

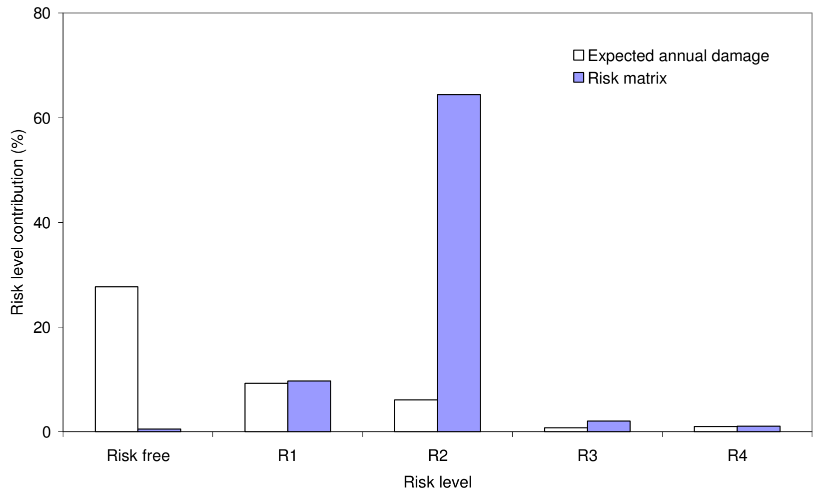

Figure 4.9 *Percentage of risk level for two risk level map: risk level map of expected annual damage, and risk level map of risk matrix*

4.3.2 Physically-based approach

The aim of the physically-based approach is to assess the flood damage associated with specific flood events under certain physical conditions, for example in case of an intentional dike break as a short-term flood mitigation measure.

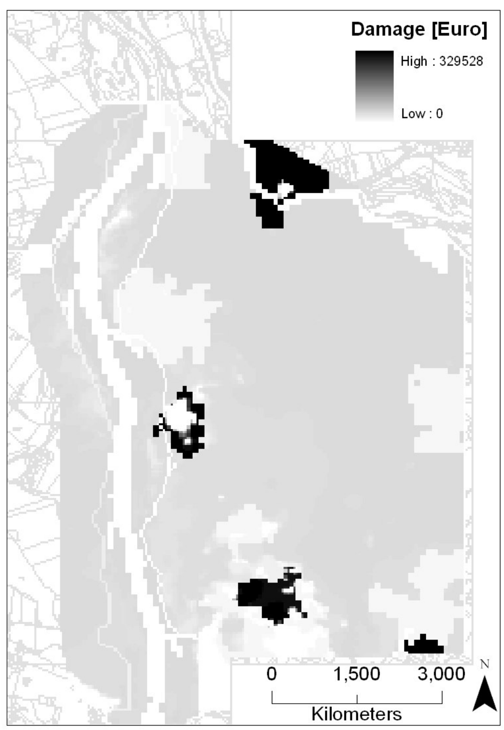

Figure 4.10 *Flood damage in Euro per 50X50m cell for 200 year flood with dike break near km 410, monetary loss*

To estimate the applicability of the physically-based approach, a 200-year flood event has been simulated at the study area. For the upstream gauge station at Tangermünde, a flood event with a 200 year return period was generated using the scaling method introduced in Chapter 3. An artificial dike break is simulated near Elbe km 410. The maximum inundation depth and maximum flow velocity maps are simulated using SOBEK1D2D. Using the physically-based approach, the economic monetary in Euro can be calculated (Figure 4.10).

The magnificence of differences in Figure 4.10 is large. Clearly, the towns of Sandau and Haverlberg are pointed out as areas that are most at risk.

The physically-based damage assessment can be used to support decision making related to short-term flood mitigation measures such as evacuation before a flood. It can also be used to assess a flood mitigation measure aimed at reducing the potential damage. This, however, requires simulations at a larger spatial scale, in order to take the non-local aspects of the measures into account (Chapter 5).

4.4 Uncertainty Analysis

To assess the contribution of uncertainties from each part of the integrated system, an uncertainty analysis is carried out following the steps of: 1) identifying uncertainty contribution, i.e. importance of parameters in the risk models through sensitivity analysis; and 2) propagating uncertainty through hydraulic models and risk models using the MCS method. In order to assess impact of implementation of measures, scenarios simulated using the MCS.

4.4.1 Sensitivity analysis

The purpose of applying sensitive analysis is to identify the significance of uncertainty contribution of each parameter involved in the risk model. According to a previous study (De Blois and Wind, 1996), the most significant uncertainty sources for FRA are (in order of significance): 1) for a river without dikes: discharge, water level, inundation damage functions, inundation depth, and land use; 2) for a river with dikes: whether there is a dike breach, discharge, water level, elevation, and damage functions.

To carry out sensitive analysis, both local SA and the Morris method are applied (Morris, 1991) (Eq. 2.4, Chapter 2). To make the analysis more focused, SA is applied to a risk model that calculates inundation depth and percentage damage according to the upstream inflow. The reference situation is set up with the design dike height with maximum correction of 1m, a inflow discharge of 3000m^3/s, and calibrated rating curve coefficients a_r and b_r (Eq. 3.3) varying along the river km. A change between -30% and 30% has been made to each parameter, one at a time. Two cases are analyzed: with dike and without a dike. Sensitivity densities of these four parameters are calculated and shown in Table 4.2.

Table 4.2 *Sensitivity density of parameters for risk model*

Parameter	Sensitivity density (%)	
	With dike	Without dike
Inflow discharge	0.33	0.33
Rating curve coefficient *a*	3.81	48.35
Rating curve coefficient *b*	3.81	48.35
Dike uncertainty	1.66	-

The result shows a significant uncertainty contribution from the rating curve coefficients, followed by the dike height correction, and the inflow discharge.

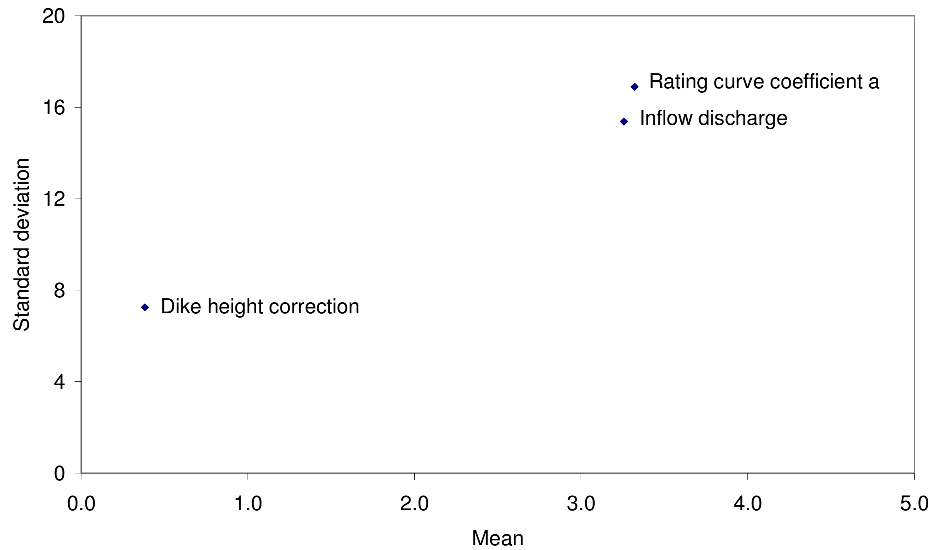

Figure 4.11 *Elementary effects of flood damage model using the Morris method*

A similar result is obtained with the Morris method (Figure 4.11). The Morris elementary effects analysis shows that rating curve coefficient *a* and *b*, as well as the inflow discharge, provides more significant effects to the model, whereas the dike correction has a smaller effect compared to the other three factors.

4.4.2 Monte Carlo simulation

It turned out to be difficult to quantify the uncertainty of the parameters in terms of a probability distribution, for example the uncertainties of the depth ~ damage curve involving both land use and elevation. The uncertainty of land use is only accounted for by the change in the hydraulic roughness coefficient (Manning *n*) between a minimum and maximum value (Chow, 1959; Shaw, 1994), but not in the depth ~ damage function. The difficulty of quantifying uncertainty also occurs in the shape of the flood damage functions, as no objective analysis of a clear statistical distribution for the damage curve coefficients had been reported. Therefore, a uniform distribution is assumed for the percentage damage for different inundation depths. The uncertainty of the damage curve coefficients is obtained by comparing different damage functions from the literature (NRC, 2000; Van der Sande, 2001; IKSR, 2001; Kok, 2001). Some studies use one to three standard deviations for the uncertainty of the damage coefficient (NRC, 2000). A comparison of flood damage functions from the literature by the author indicates approximately a 30% difference in percentage damage. Therefore, a 30% uncertainty is assumed for the damage curves coefficient. An arbitrary uncertainty of 30% in the rating curve coefficients is also assumed. The uncertainties of all key parameters are listed in Table 4.3.

For practical reasons, the more efficient stratified Latin Hypercube Sampling (LHS) method is used for the UA (Saltelli et al., 2000). To avoid long computing times, a screening method is applied first for the hydrodynamic model. The most sensitive parameter - the roughness - is changed, one step at a time, to generate a range of maps of the maximum inundation depth and maximum flow velocity. MSC is then applied to the risk models with the uncertainties propagated through hydraulic models. Finally, the uncertainty distributions are determined for the relevant indicators for risk, namely the percentage damage, the monetary damage and the risk levels.

Table 4.3 *Uncertainty identification of model parameters*

Parameters	Mean	Std	Min	Max	Statistical distribution	Uncertainty
Discharge [m^3/s]	1830	671	150	6000	Gumbel	-
Dike height correction [m]	1	0.3	0.1	2.0	Uniform	1std
Damage Coefficient (*d*)	Mean(*d*)	Std(*d*)	0	1.0	Uniform	0.3*d_h
Rating curve Coefficients (a, b)	Mean(*a,b*)	Std(*a,b*)	Min(a,b)	Max(a,b)	Uniform	0.3**a(b)*
Gumbel Coefficients (a, b)	Mean(*a,b*)	Std(*a,b*)	Min(a,b)	Max(a,b)	Uniform	0.3**a(b)*

Note: *std* indicates standard deviation, d_h denotes the damage coefficient at inundation depth *h*.

To demonstrate how risk assessment benefits from information on uncertainty, uncertainty analyses have been carried out to investigate the significance of the dike for flood risk. Scenarios have been set up for the UA using the statistical risk model: with and without a dike. It has to be noted that using the traditional distribution plot, confusion is caused by the overlap area of these two scenarios. A scatter plot can be used to distinguish the scenarios (Figure 4.12).

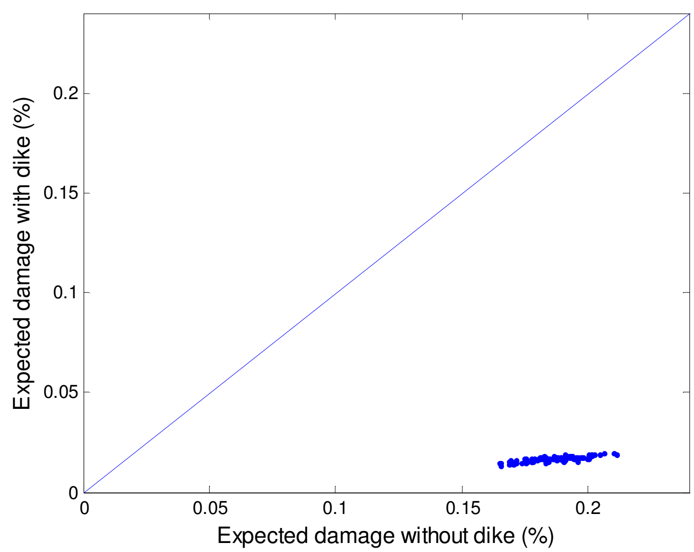

Figure 4.12 *Scatter plot of the annual expected damage (%) of scenarios of dike effect and without dike*

Figure 4.12 shows a cloud of the results of UA using Monte Carlo simulation. In agreement with what has been found in Figure 4.6 and Figure 4.7, the results point out a more conclusive significance of the dike effects: the expected damage is significantly higher when there is no dike protection. This conclusion seems obvious, but the underlying idea is to present quantitative risk distribution for different scenarios. For example, to present the difference in risk of any defense system compared to the situation with different flood mitigation measures, such as heightening the dike, deepening the river channel, or shifting the dike separately, which would result in a change of the rating curve coefficients in the risk equations (Eq. 3.2). The main purpose of a scatter plot is to illustrate whether the effect is statistically significant.

Furthermore, uncertainty analyses for the dike break effect using the physically-based approach, have been carried out. The study is set up for two cases, with and without a dike break at the right hand side of the dike near Sandau area at km 400.

Using SOBEK1D2D, three flood events, associated with a return period of 28-years, 50-years and 200-years, are simulated. An UA of the hydraulic model is carried out for the most sensitive parameter, the roughness value. It has been changed between a minimum, nominal and maximum value associated with ten reclassified land use types. In total, sixty simulations were carried out, providing maps for the maximum inundation depth and maximum flow velocity, for the situation with and without an artificial dike break, respectively. These hydraulic results, together with the samples of flood damage function coefficients, are used as input for the Monte Carlo simulation of the physically-based approach.

Figure 4.13 shows a scatter plot for the uncertainty in the average flood damage for these two scenarios. Another obvious result is found, that the damage is always higher when a dike break happens. This observation, however, only pertains to the local effect. Non-local effects consist of the reduced downstream water levels, a delay of the flood peak due to the temporal retention storage, as well as the reduction of the flood damage downstream. To obtain a non-local effect, it requires a hydraulic simulation and risk assessment for a larger area covering the area of interests.

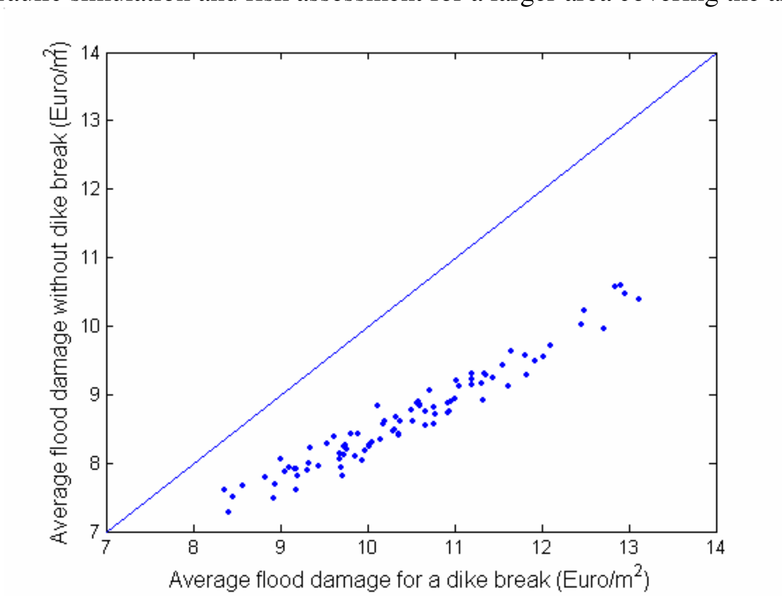

Figure 4.13 *Scatter plot of economic loss for scenarios with an artificial dike break and without a dike break*

Unlike the statistical approach, the physically-based approach can represent the instant physical change associated with a certain flood event. Therefore, to capture the short-term effect of any flood mitigation activity, it is recommendable to carry out an UA using a physically-based risk model first.

4.5 Conclusions

A comparison has been made of the applicability of the statistical and physically-based approach at the local scale in the context of IFRA.

For long-term planning, a statistical approach can be followed to identify the area vulnerable to flood risk. It can be used to provide decision makers with qualitative information on the effects of the construction of for example a dike or other structural flood defense works. The aim of this approach is to determine the annual expected damage from the discharge probability distribution and damage resulting from events with a different probability of consequence ($risk = probabilty \times effect$). It is best combined with a steady flow hydraulic model like HEC6.

This approach, however, cannot account for the effects of flow velocities on the flood risk, recognized to play a significant role (Figure 4.8). The alternative is to use a qualitative indicator, based on a matrix combining the effect of flood damage and (maximum) flow velocity on the risk level. Although subjective, the advantage is that more factors can be included in the analysis, and river managers can define the risk levels according to their preferences and experience. Comparison of the risk maps for the Sandau region shows a different spatial pattern of the risk level map when flow velocity is taken into account.

A physically-based risk assessment serves short-term planning purpose, which aids decision making for operational activities such as emergency evacuation. It can be used to assess the effects of flooding resulting from an intentional dike break, caused to protect downstream areas more vulnerable to damage, for example. By determining the effect for a particular flood event the most appropriate location for the dike break can be chosen. Contrary to the statistical approach, the physically-based approach is best combined with a hydrodynamic model like SOBEK1D2D. Application to the Sandau region for a dike break with a 200-year flood event leads to a damage pattern, in which the residential areas of the towns of Havelberg and to a lesser extent Sandau are clearly pointed out. It should be noted that, although the risk matrix could be combined as indicator with the physically-based approach, the risk matrix should be used with caution. The reason is that the velocity index used in the risk matrix is obtained from a statistical approach, i.e. normalized value presented in Chapter 3.4.3, which does not correspond to the actual flow velocity (map) obtained for each flood event.

UA has been found beneficial to determine how significant the differences in effects of different flood management measures are. For the statistical approach the risk, expressed in terms of the expected annual damage, with and without dikes is compared. Obviously, the resulting scatter plot shows a lower risk for the case with the dike present, but the additional value lies in the uncertainty distribution, which indicates the difference is also significant, compared to the uncertainty range. Similarly, an UA is carried out for the physically-based approach, with a scenario with and without dike break. Here too the resulting scatter plot demonstrates that the distinction between the two scenarios is statistically significant.

Chapter 5

Comparison of the Hydrodynamic and GIS approach

As explained in Chapter 3 and demonstrated in Chapter 4, the physically-based risk assessment is more suitable for short-term planning, for example to decide where to induce an artificial dike break in order to protect downstream, economically vulnerable areas. As explained in Chapter 2, the selection of a hydraulic model should be carried out with care. In principle the physically-based approach requires a hydrodynamic model in view of the event-based orientation, but this still leaves open the issue of how complex such a model should be. In the case of a dike break a full dynamic model (for example SOBEK1D2D) can be used to determine the maximum inundation depth and flow velocity in the flooded areas. A recent development is the emergence of rapid, GIS-based approaches for inundation modeling which take a flood event in the main channel of the river as starting point. In this Chapter such and approach is used and extended with an approximation scheme to include the effect of flow velocity in the inundated areas.

The IFRA framework proposed in Figure 3.9 allows for both the GIS-based approach and hydrodynamic approach for inundation modeling. Both approaches are compared in this chapter for a simulated dike break near the town of Sandau. Results are obtained at the local scale in terms of the maximum inundation depth and flow velocities in the area behind the location of the dike break, and at the non-local scale in terms of the reduction of water levels in the main channel and flood damage that has been avoided.

Uncertainty analyses are carried out to examine the difference in significance of two uncertainty sources: the aggregation method for the elevation data, and the hydraulic roughness.

5.1 Introduction

A dike break can be either a natural event, or an intentional measure. A natural dike break can occur when the pressure is too high due to the high water level, whereas an intentional dike break is a man-made deconstruction of the dike at a specific location. Nevertheless, an artificial dike break can reduce the water level downstream, and hence reduce the flood risk. For the Elbe river, it has been found that flood mitigation measures such as the use of a retention basin through an intentional dike break, has significantly reduced the flood risk at downstream areas during the flood in 2002 (DKKV Publication 29e, 2004). On 22 August 2002, the polders along the Havel River (a tributary of the lower Elbe), were used for an intentional destruction of the dike. The temporary storage of flood water in the polders reduced the water levels up to 66 cm at Wittenberge (km 454.6), a downstream gauge station of Elbe River after the joint of Havel (DKKV publication 29e, 2004). This demonstrates the usefulness of an intentional dike break as a short-term measure to mitigate flood risk. This type of operation can be carried out in economically less vulnerable areas. On the other hand, an unprovoked, "natural" dike break may cause substantial risk both in terms of economic loss and life at risk, depending on the location of the dike break.

For these reasons it was decided to examine the consequence of a dike break along the Elbe river, for incorporation into the pilot DSS. Figure 5.1 schematizes the general approach for the simulation of a dike break. The hydrodynamic model SOBEK1D2D (Stelling and Duinmeijer, 2003) is employed to simulate the artificial dike break, which produces maximum inundation depth and flow velocities at the flooded are behind the dike. The inundation damage in terms of Euro is then calculated using the proposed damage function from Elbe_DSS. The results include the maximum inundation depths, the flow velocities, and the flood damage in flooding areas. In view of the reduction of the downstream flood risk, non-local impacts (i.e. reduced water levels) are included in the analysis as well.

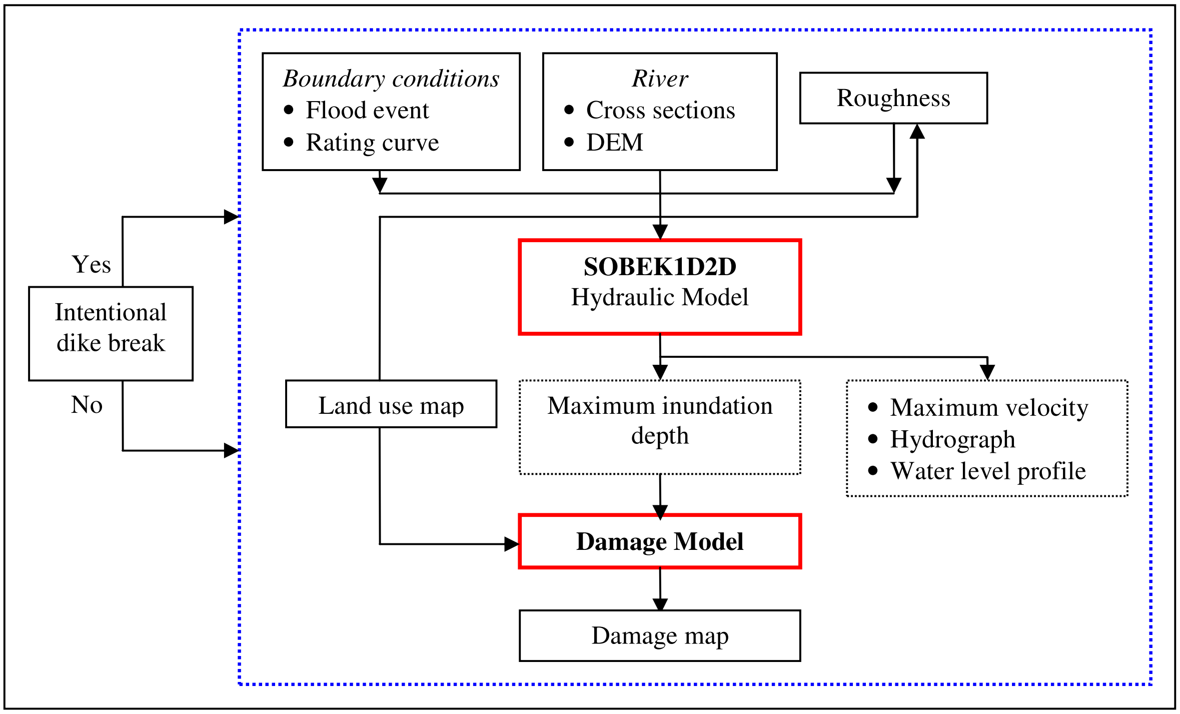

Figure 5.1 *Outline of the hydrodynamic modeling of a dike break. To give the impact of dike break, the modeling is compared with the result of simulation without a dike break*

5.2 Data Conditions

5.2.1 Modeling area

To estimate how effectively flood mitigation measures can reduce the risk at downstream the city of Wittenberge, as example a dike break is simulated between two boundaries of Tangermünde and Wittenberge (Elbe km 388-km 454.6) (Figure 5.2, next page).

Due to the lack of dike height data and different sources of DEM, pre-processing has been carried out to improve the consistency of the data. The following sections present the processes of data pre-processing.

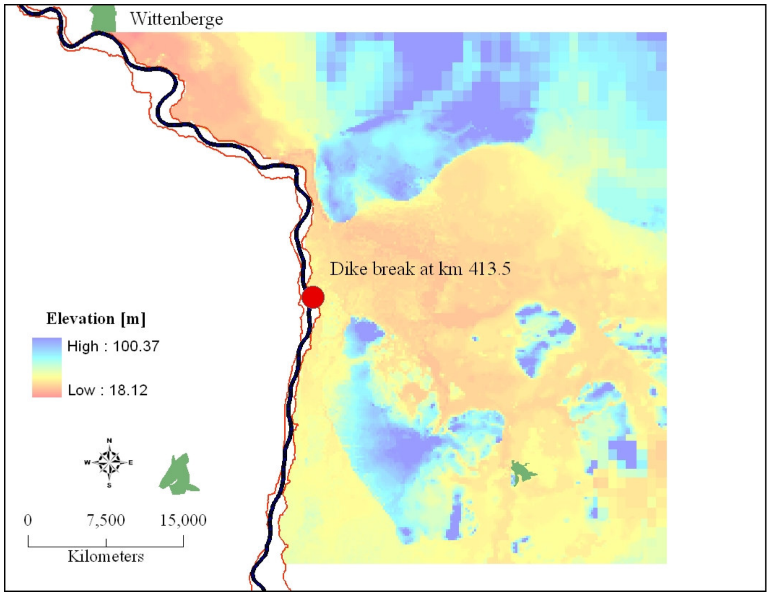

Figure 5.2 *Dike break modeling area, Sandau between km 388.2 and km 450, with the location of the dike break near Sandau*

5.2.2 Dike height

A dike line provides geo-information of the dike along the river horizontally, and must be complemented with information on the dike height. The dike line and dike height data have been digitized following two major steps (Jankiewicz et al., 2005):

- *Source variability*: The dike data were collected from all sorts of resources (e.g. IKSE, 2001) and formats, and has been put into a table by river kilometer. It should be noted that the original dike height data were available in different formats such as tables, figures, topographical maps and text descriptions with dike height information. These data are available in different datum systems, namely the DHHN92 - the Deutsches Haupthöhennetz 1992, and NHN - the Normalhöhennull.
- *Interpolation*: During the dike preparation the raw ASCII files with dike height data had to be transferred into a GIS shape file following the polyline of dikes. The location of each data point is identified from the intersections between the dike lines and the cross section lines. This procedure introduces additional uncertainty due to the geographical uncertainties caused by the re-projection of dike line and the cross section lines, apart from direct interpolation of the dike height data.

In summary, the dike height data are prepared and processed as the elevation above sea level, and associating with the 1D river channel chainage indicated in terms of kilometers. Uncertainties can be attributed to the inhomogeneity of data sources, and the interpolation process. The occurrence of missing dike data was found to be a key problem to be dealt with prior to the dike break simulation. Gaps in the dike height data occurred along both sides of the river.

To complete the dike height data, a two-step interpolation is followed. The interpolation is taking place in one dimension, using the river kilometer as reference parameter, and a 2nd order polynomial:

$$H_r = 0.0001x^2 - 0.2776x + 123.736 \quad (5.1)$$

$$H_l = 0.0001x^2 - 0.2801x + 124.29 \quad (5.2)$$

where the subscripts r and l denote the right-hand side and the left-hand side respectively, and x is the kilometer length along the river. R^2 is 0.9969 and 0.9988 for the right and left dike, respectively.

This interpolation is essential to complete the cross section boundaries, as well as the 1D risk damage approach. The results are shown in Figure 5.3 a, b.

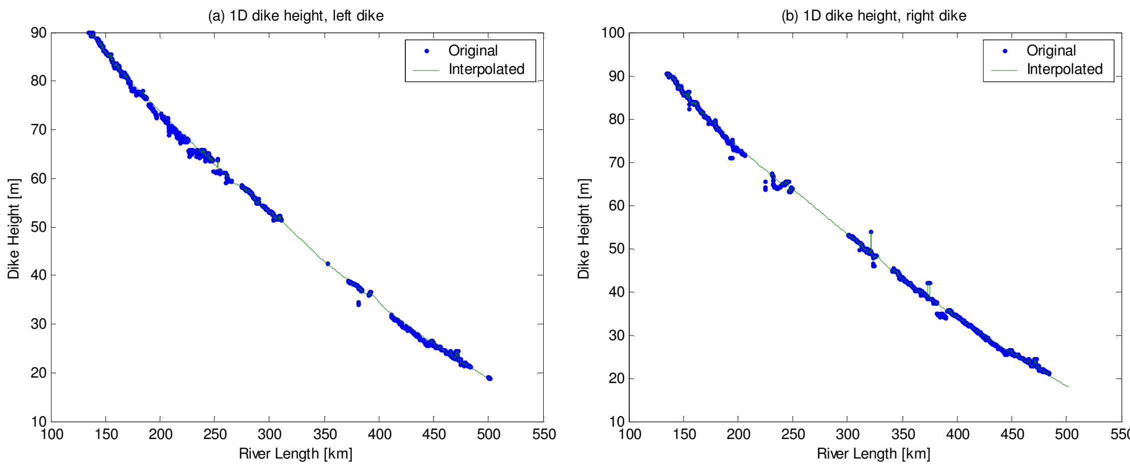

Figure 5.3 *1D interpolation of dike height*

Next, 2D interpolation was necessary to merge the dike height data correctly with the DEM data, based on the "closed" dike line's coordinates as most of the open parts is the junction from tributaries to the main river channel. Values from the dike height table are used when available, followed by linear interpolation between nearest available dike height points. The next step is to compare the interpolated or available dike height value with the elevation data, and the maximum value is assigned to the dike cell as the final dike height. Finally, the interpolated dike data are merged with the digital elevation data, which will be used for overland flow computations for the dike break simulation.

5.2.3 Digital elevation model

Two sources for the elevation data are used in the dike break simulation. The basic DEM, LGB2004, is originally of 5 m horizontal resolution, and covers mainly the main channel of Elbe River (LGB, 2004). These data cover the main channel of the river basin including the floodplains, and area behind the dike. The second source of elevation data is 1000m resolution GTOPO30, a geographic database which is developed at the U.S. Geological Survey's (USGS) EROS Data Center, providing comprehensive and consistent global coverage of topographically derived data sets (LPDAAC, 2004). These data are distributed by the Land Processes Distributed Active Archive Center, located at the U.S. Geological Survey's EROS Data Center http://LPDAAC.usgs.gov. For the dike break simulation these data are used to complete gaps in the basic DEM for the overland flow simulation. The use of two difference sources of data introduced additional uncertainty. The difference in accuracy between the LGB2004 and GTOPO30 data might be reflected by the decimal accuracy the two data reaches, which is to 0.01m and 1.00m, respectively.

In addition to uncertainty of different data resources, for DEM data, uncertainty also comes from the aggregation methods used to obtain larger resolution size from finer resolutions. As a reasonable scale from both the practical and the computational point of view, 100m resolution was adopted for the DSS. Thus, the 20m resolution DGM20 is aggregated into 100m. Three types of aggregation methods are compared. These are: aggregation using the minimum value, the mean value, and the maximum value of the neighbor cells.

Table 5.1 lists the statistical parameters of the DEM after aggregation near Sandau area for three aggregation methods. It turns out that the different aggregation methods cause a deviation between 0.30m and 0.65m.

Table 5.1 *Statistical parameters depending on the aggregation method chosen for the DEM*

Elevation Data		Elevation (m)			
		Minimum	Mean	Maximum	Std dev.
Elevation data after aggregation	Minimum aggregation	18.02	31.65	94.01	8.02
	Mean aggregation	18.11	32.11	100.4	8.38
	Maximum aggregation	18.12	32.6	107.59	8.81
Original elevation data		18.02	31.95	107.59	8.10

Another uncertainty comes from *sinks*. In ArcGIS a sink is a cell or set of spatially connected cells whose flow direction cannot be assigned to one of the eight valid values in a flow direction grid. This can occur when all neighboring cells are higher than the processing cell, or when two cells water flow into each other creating a two-cell loop. These errors are often due to sampling effects and the rounding of elevation data to integer numbers. Naturally occurring sinks in elevation data with a cell size of 10 meters or larger are rare on the floodplains (Mark, 1988) except for glacial or karst areas, and generally can be considered errors.

To create an accurate representation of flow direction and, therefore, accumulated flow, it is best to use a dataset that is free of sinks. A DEM that has been processed to remove all sinks is referred to as a *depressionless* DEM. The identification and removal of sinks, when trying to create a depressionless DEM, is an iterative process. When a sink is filled, the boundaries of the filled area may create new sinks which then need to be filled.

Table 5.2 shows the difference between the elevation data before and after filling the sinks by taking the average of the neighbor cell's value, when the difference between two neighbor cells is at least 10m. The difference is on average 0.16m lower before the sinks are filled.

Table 5.2 *Statistical parameters of the DEM with and without filling of sinks*

Elevation / Filled DEM	Minimum (m)	Mean (m)	Maximum (m)	std dev (m)
Mean 100m cell size	18.11	32.11	100.4	8.38
Filled mean-100m cell size (10m)	18.12	32.27	100.4	8.32
Difference (non-filled – filled)	-0.01	-0.16	0.00	-0.06

5.2.4 Flood damage functions

The assessment of the flood damage is based on functional relationships between the inundation depth and the flood damage as a percentage of the maximum potential damage, specified by land use type. As these functions are case dependent these had to be derived for the study area. Based on the Rhine Atlas (Rhine Atlas, 2001) and the difference between the East- and West-German economy, the Grossmann (2004) approach has been followed.

In this approach, seven aggregated land use classes are obtained by reclassification of the 44 CORINE land use classes (Appendix I). For each land use class, the total damage consists of three components: the damage to immobile assets, the damage to mobile assets, and profit-loss related damage (Grossmann, 2004; De Kok and Huang, 2005). For each damage component the percentage damage is obtained from the inundation depth using the functions shown in Table 5.3.

Table 5.3 *Rhine Atlas flood damage functions: damage (%) ~ inundation depth (h)*

Land use class and component	Damage (%)
Residential Immobile	$2h^2 + 2h$
Industry Immobile	$2h^2 + 2h$
Traffic Immobile	$\begin{cases} 10h & h < 1m \\ 10 & h \geq 1m \end{cases}$
Residential mobile	$11.4h + 12.625$
Industry mobile	$7h + 5$
Traffic mobile	$\begin{cases} 10h & h < 1m \\ 10 & h \geq 1m \end{cases}$
Agriculture grassland immobile	1
Cultivated agriculture immobile	1
Agriculture grassland profit	50
Cultivated grassland profit	50
Forest	1

To obtain the monetary loss, the property values for the Rhine are (ICPR, 2001) are translated to the Elbe economy by comparing the East- and West-German economies (Grossmann, 2004). Table 5.4 shows the property density in Euro per m^2 for each aggregated land use type.

Table 5.4 *Property density values for the property components for the Elbe river basin (Grossmann, 2004)*

Land use class	Immobile (Euro/m^2)	Mobile (Euro/m^2)	Profit (Euro/m^2)
Residential	145	12	0
Industry	25	2	0
Traffic	25	2	0
Cultivated agriculture	7	0	0.1
Agriculture grassland	7	0	0.1
Forest	0	0	0.025
Other	0	0	0

For each cell the total damage in euros can be determined from the cell's land use and the calculated inundation depth by combining Tables 5.3 and 5.4.

5.3 Dike Break Modeling Using SOBEK1D2D

To assess the effectiveness of the measure of a deliberate dike break for the purpose of risk mitigation, the area upstream of city Wittenberge is selected for the dike break simulation. The hydraulic model is set up for Elbe is km388.2 – 456, with a discharge time series as upstream boundary, and the rating curve (Otte-Witte et al., 2002) as downstream boundary. The intentional dike break takes place at an upstream area of low economic importance near km 413.5, at the right hand side of the river.

The hydrodynamic modeling software SOBEK1D2D (Stelling and Duinmeijer, 2003) is employed to simulate the dike break scenarios. For the cases with and without a dike break, or with a dike break at different locations, the boundary conditions are identical. River cross-profiles data and elevation data are used to model the channel flow and overland flow, respectively. The channel roughness is calibrated with historical time series of 1960-1995 (Helms et al., 2002a, b). For the area at risk (where the 2D overland flow occurs), the roughness is converted from land use using the Manning values (Appendix IV). For the modeled dike break the river flow is diverged into the area behind the dikes. The maximum inundation depth and maximum flow velocity can then be calculated for the flooded area with the overland flow module in SOBEK1D2D. Hydrographs for the channel water level and discharge are also obtained. The maximum inundation depth and land use map are the input for the damage model (see Section 5.3), from which a damage map is calculated using the damage functions.

The possible economic effect of the dike break is obtained by comparing the resulting damage for the situation with a dike break at the area downstream. The hydraulic impacts are indicated by the hydrographs at downstream boundary, consisting of the lowering of the water level against time, i.e. a water level (discharge) hydrograph, or against the location, i.e. longitudinal water level profiles.

5.3.1 Dike break simulation method

In SOBEK1D2D, a dike break can be modeled either in one or in two dimensions using the *Flow Dam Break Reach (FDBR)* option of SOBEK1D2D. With the flow dam break reach option, the dike break can be set up with a specific growth rate of the break in both the vertical and the horizontal direction. The horizontal development is not related to the chosen grid cell size, but can have any value (Appendix V).

The FDBR option comprises a 1D reach in which SOBEK accommodates an artificial weir. The end of a FDBR is connected to a 2D grid cell by linking two 1D connection nodes. The discharge through the FDBR is computed using the structure equation of the weir (Verheij, 2002), while taking into account the actual crest level and crest weir as well as the water levels at both ends of this reach. SOBEK1D2D sets the elevation of this 2D grid cell equal to the lowest user-defined elevation of the breach.

A dike break is simulated in two phases. Starting from a certain moment, the gap crest level is going down with a constant gap width. When a certain maximum depth of the gap is reached, the width of the gap starts to increase. A detailed description of the simulation of the dike break using the formula of Verheij-vdKnaap 2002 (Verheij, 2002) can be found in Appendix V.

5.3.2 Determine dike break time and width

Various tests have been carried out to determine how wide the dike break should be and when it should be broken. The tests point to a significant water level reduction when the maximum dike break width falls in the range of 150-200m (Figure 5.4).

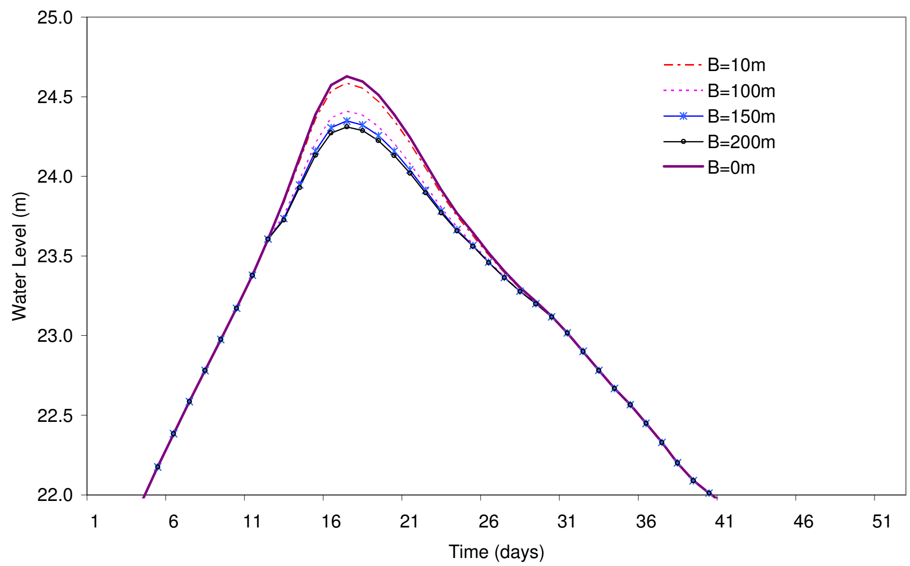

Figure 5.4 *Water level at the gauge station Wittenberge associated with different maximum dike break width*

A time of 20 minutes for full development of the dike break is finally determined using the same trial-and-error approach, as normally a real intentional dike break can develop rapidly. This value is area-dependent due to the flow and DEM characteristics. Finally, a width of 200m is selected as the maximum dike break width. Some calculation of how to determine the dike break timing and width can be also found in the MSc thesis from Abazi (2005).

5.3.3 Results

A dike break is simulated for a 200-year return period flood, denoted as Q200, at the upstream boundary. A 100 m resolution is used for the overland flow (2D) set up of the model using the processed DEM. Ultimately a dike break width of 200m and full vertical development time of 20 minutes, are adopted.

Two scenarios are simulated: 1) a dike break at km 413.5 at the right hand side of the river, and 2) dike break at km 431.3 at the right hand side of the river. The first dike break is the intentional dike break aiming for damage reduction at downstream area, as the area near the town of Sandau (near Elbe km 413.5) can be considered to be of lower economic importance compared to the city of Wittenberge downstream (near Elbe km 431.3). The second dike break aims to simulate a failure of the dike downstream, as the reference case to evaluate the effectiveness of the dike break measure as flood risk mitigation of Wittenberge.

Several indicators are used to present the flood risk and characterize the effect of the dike break measure: the maximum inundation, the maximum flow velocity, flood damage, and the reduction of the water level downstream of the dike break.

The inundation map (Figure 5.5) shows a maximum inundation depth of 4.98m at the flooded area, where the maximum flow velocity (Figure 5.6) reaches 2.22 m/s. Both maps can be regarded as risk maps in terms of hydraulic characteristics. These two maps show different patterns.

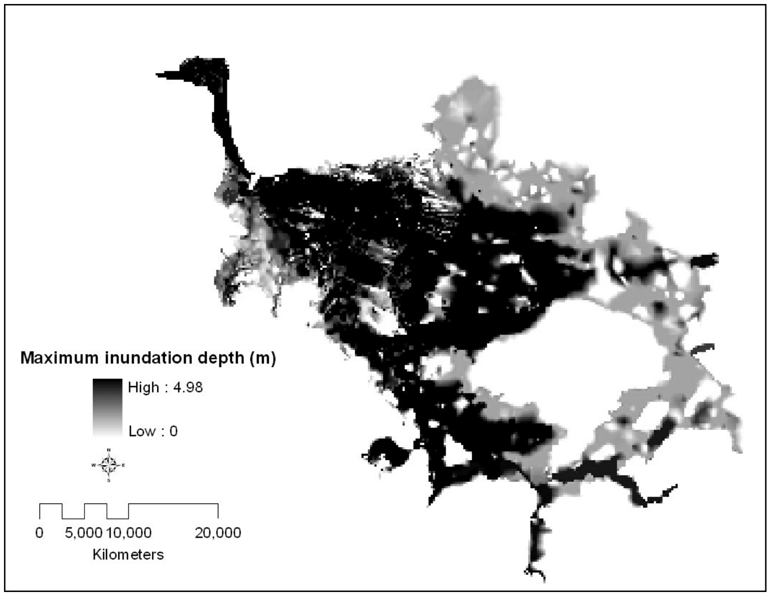

Figure 5.5 *Map of maximum inundation depth for dike break at km 413.5*

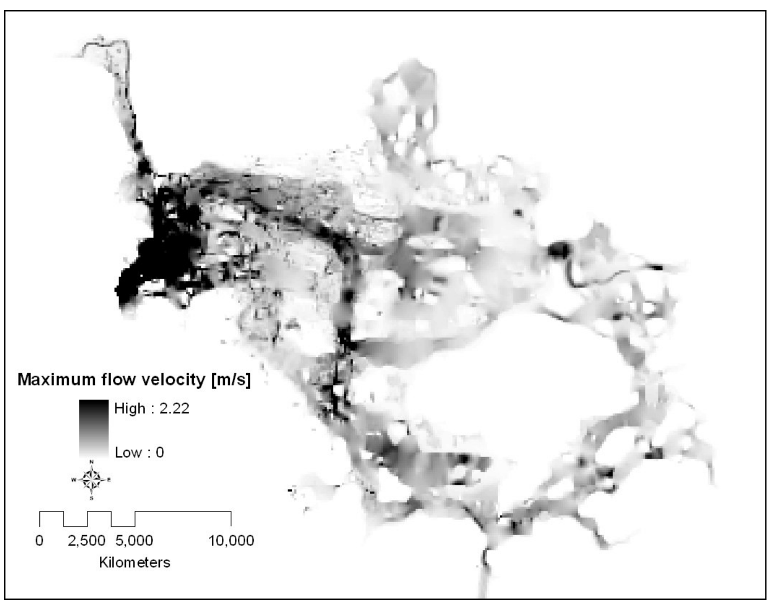

Figure 5.6 *Map of maximum velocity for dike break at km 413.5*

The inundation damage for each cell is calculated using the depth ~ damage function (Table 5.3 and Table 5.4). The intentional dike break at km 413.5 results in a total flood damage of 132 million euro and 2,418 km^2 flooded area. The damage map is showed in Figure 5.7.

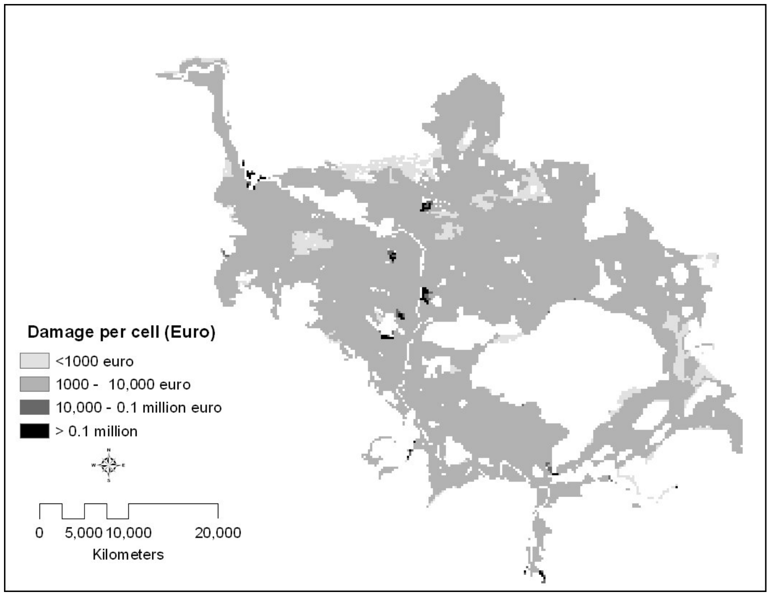

Figure 5.7 *Map of damage for dike break at km413.5*

The aim of an intentional dike break at km413.5 is to mitigate risk/damage downstream near the city of Wittenberge. The dike break at km413.5 with a 200m width and full growth in 20 minutes at day 17 of the flood event (five days before the peak), results in a reduction of 40 cm of the channel water level near Wittenberge, compared to the situation without dike break (Figure 5.8).

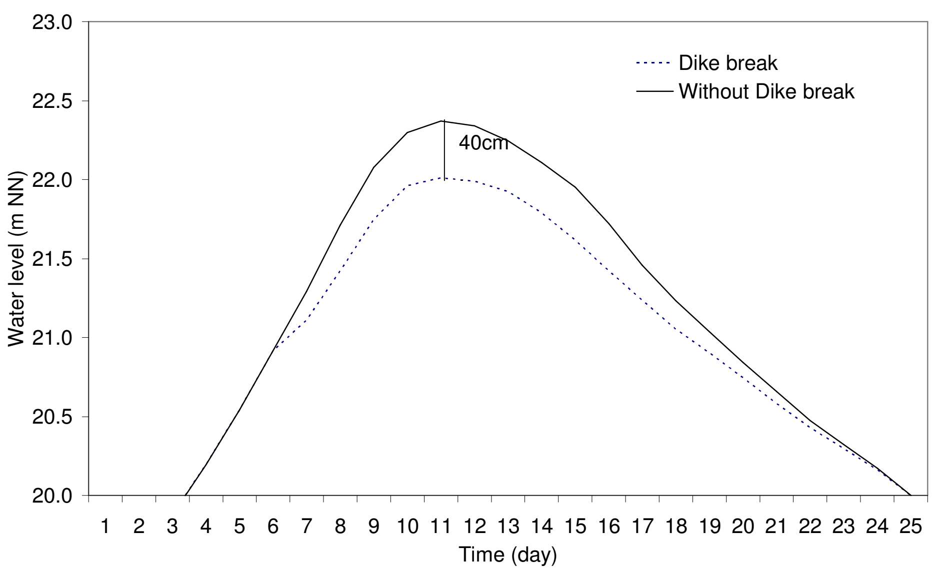

Figure 5.8 *Reduction of 40 cm water level at gauge station Wittenberge (Elbe km450) with dike break at km 413.5*

The statistics obtained from the comparison of two scenarios (Table 5.5): dike break at km 413.5 and dike break at km 431.3, also shows effective risk mitigation by an intentional dike break: with a dike break at km413.5, the total damage is reduced from 420 million euro to 132 million euro, and the flooded area is reduced from 3778 km^2 to 2418 km^2.

Table 5.5 *Dike break results*

Scenario	Inundation depth (m)		Velocity (m/s)		Cell damage (Euro)		Total damage	Area
	mean	max	mean	max	max	mean	(Euro)	(km^2)
km413.5	0.25	4.98	0.07	2.22	948566	2271.6	132 million	2418
km431.3	0.53	8.52	0.14	1.98	1557271	7578	420 million	3778

5.4 Inundation Modeling Using GIS Approximation

The maximum inundation depth can be approximated using the water volume flowing into the area behind the dike. This means that GIS technology, combined with river flow routing using a 1D hydraulic model, can be an alternative to approximate inundation modeling.

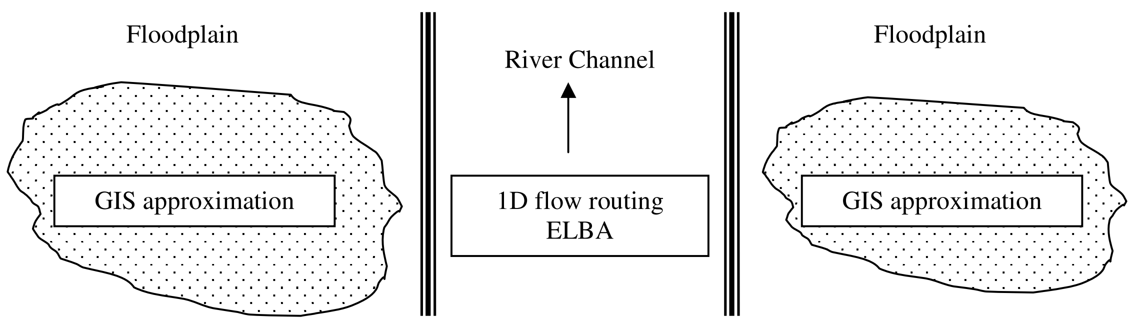

Figure 5.9 *Conceptual framework for 1D inundation approximation using GIS technology*

Figure 5.9 schematizes the conceptual framework for such an approximation. Relevant model components include the slope of the basin, as well as the boundary of each sub-basin using GIS hydrology tools. The following sections present each of these components in detail.

5.4.1 General approach for the GIS approximation

GIS approximation stands for the combination of flood routing in the river channel with ELBA model, which is a 1D approach, and the inundation map is produced using DEM and GIS hydrological tools such as flow accumulation tool, and tool to generating watersheds (Figure 5.10).

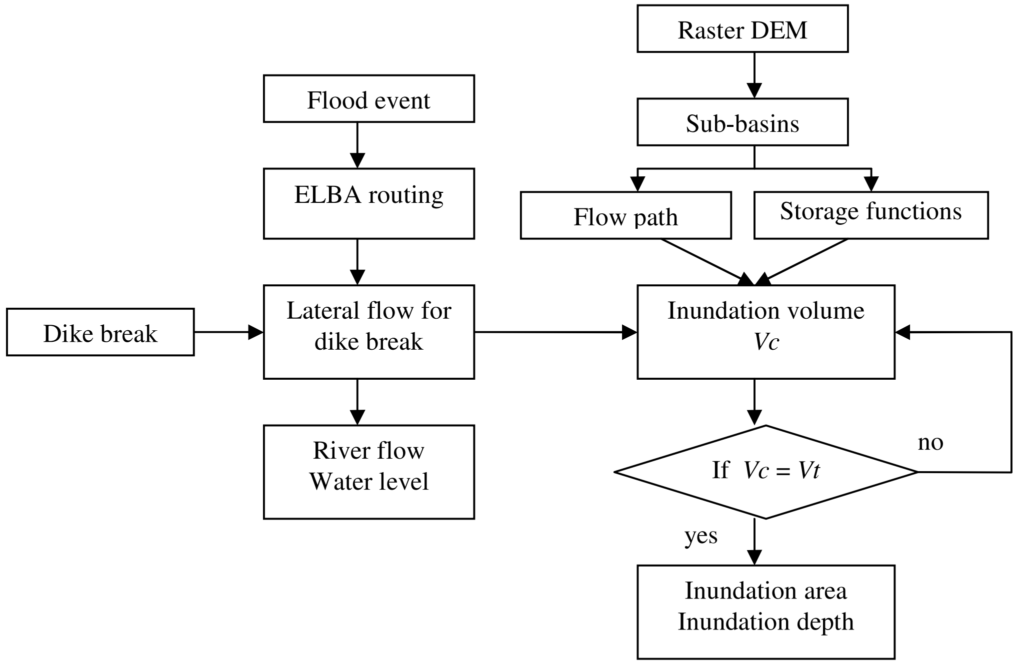

Figure 5.10 *Flowchart of the GIS approximations*

As illustrated in Figure 5.10, using the DEM raster data and inflow discharge to the river, GIS-based approach approximates the inundation depth behind the dikes following six steps, *assuming water level is horizontal in the inundated area*:

1. *Determine the sub-basins* for the modeling area. A set of sub-basins are determined following the flow direction map which is calculated using GIS hydrology tool, with the principles of that water flows to the deepest point among neighbor cells.
2. *Calculate the water level (h) ~ volume (v) curves* for each sub-basin. In the $h \sim v$ curve, the water level starts with the minimum elevation of each sub-basin and stops at the maximum elevation of the sub-basin. The volume is calculated using the expression:

$$V_i = \sum_{i=1}^{n} \sum_{j=1}^{m} (H_i - z_j) * A_j \tag{5.5}$$

   where V_i is the volume associated with water level H_i for the whole sub-basin, n is the number of water level intervals, m is the number of cells which has the elevation z_j (lower than water level H_i). A_j is the area of cell j.
3. *Find the flow path*: for each sub-basin, the neighbor basins and corresponding minimum elevation are determined. These are used to determine the flow path (or flow direction). That is, when the water level in the basin reaches the minimum level leading to connection with the neighbor basin, the water will flow into this neighbor basin.
4. *Calculates diverged water volume*: using the same flow time series of dike break calculated using the so-called Verheij-vdKnaap 2002 expressions (Verheij, 2002) (Appendix V), the volume of water for the dike break is calculated from:

$$V = \sum_{i=1}^{n} \frac{Qdb_i + Qdb_{i-1}}{2} * 24 * 3600 \tag{5.6}$$

where V is the volume of water for dike break; Q is the flow discharge in unit of m^3/s; *db* denotes dike break, n is the number of time steps in unit of days for the flood event.

5. *Determine flooded area:* the volume calculated in step 4 is taken as the flood volume (V_t in the flowchart in Figure 5.10). At the location of the dike break, an iterative procedure is followed to fill the basins and its neighbor's basins until the maximum volume (V_c) reaches this flood volume.
6. Using the result of step 5, an inundation map is finally calculated.

5.4.2 Flood routing - ELBA

To rout a flood event downstream along the main channel the German Federal Institute of Hydrology developed the translation-diffusion model ELBA (Fröhlich, 1998; Busch et al., 1999). The model has been calibrated for seven sections along the Elbe River (Appendix VI). In the model three discharge regimes are distinguished, which can be superimposed. The system function for the routing procedure is given by:

$$q(t) = \frac{L}{2t\sqrt{\pi D_h t}} \exp\left(-\frac{(ut-x)^2}{4D_h t}\right) \tag{5.7}$$

where,

L	Length of the modeled river section [km]
u	Translation coefficient [km/hr]
D_h	Diffusion coefficient [km^2/hr]
t	Time step [hr]

The parameters D_h and u have been determined for seven river sections in the trajectory downstream of Dresden for three discharge regimes (Helms et al., 2002a, b). The ELBA model has been developed and tested using the historical data of Elbe daily discharge from 1960-1995.

The ELBA routing only concerns the 200-year return period flood event from the gauge station at Tangermünde to Wittenberge. With a dike break at Elbe km 413.5, the water level shows 35 cm (which was 40 cm in SOBE1D routing, Figure 5.8) reduction compared to the water level without a dike break.

5.4.3 Inundation depth

A comparison has been made between the results of SOBE1D2D and GIS approximation (Table 5.6). The results show a larger inundation depth (0.34m) and total monetary damage (20%) using SOBEK1D2D, while the total inundated area for both methods is almost identical. The reason is that the larger inundation depth in SOBEK1D2D happens at areas of higher economical values (such as urban area), which results in a larger monetary damage for each cell, and in turn the total damage.

Table 5.6 *Results for the inundation simulation, SOBEK1D2D vs GIS-based approach*

Model	Total Damage (Euro)	Damage (Euro)		Total inundated area (km^2)	Inundation depth (m)	
		mean	max		max	mean
SOBEK1D2D	76.5 million	1842	0.24million	366.73	4.98	1.36
GIS-based	59.4 million	1866	0.95million	365.32	4.61	1.02

Another reason of such a difference might be in the accuracy of the elevation data. For SOBEK1D2D directly aggregated elevation data are used without the filling procedure described in Section 5.2.3. For GIS-based approach, in order to obtain a distinction between the sub-basins, the sinks in the DEM are filled using the mean value of the neighbor cell's with the threshold value of 10m, whereas the DEM used for SOBEK1D2D didn't consist such a filling effect. Thus, the sinks in the DEM used for SOBEK1D2D may produce locally larger inundation depths, which in turn result in larger monetary damage.

Nevertheless, GIS approximation clearly agrees with SOBEK1D2D in terms of the inundation area as well as damage distributions. This can be seen in the comparison of two damage maps (Figure 5.11). The two classified maps are identical, except the difference in the maximum cell damage.

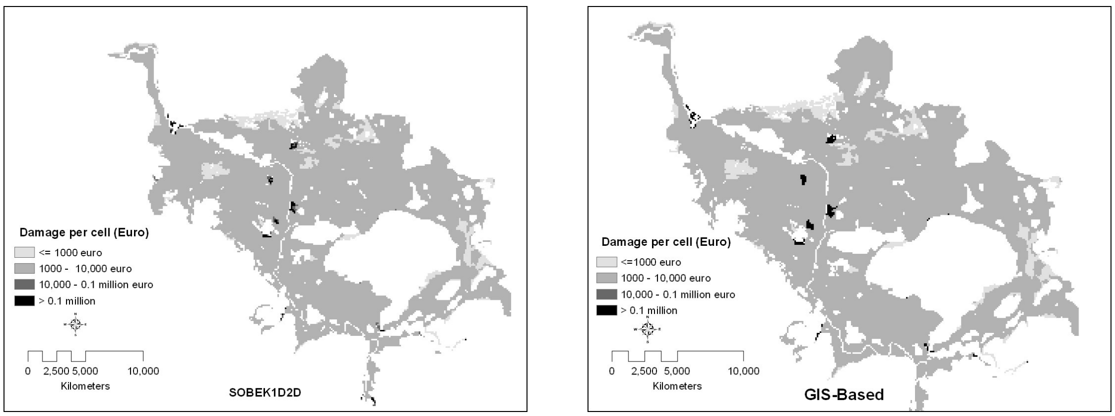

Figure 5.11 *Flood damage for the dike break at Elbe km 413.5 using SOBEK1D2D and GIS-based approach*

5.4.4 Flow velocity

For rapid assessment, in principle, a GIS-based approach can be used to approximate the flow velocity based on the geographical information of the DEM, if the *water surface slope* is approximated reasonably. The assumption can be made that the flow takes place in the direction of the steepest water surface slope. Therefore, a water surface slope map for the modeling area is necessary first.

However, such a slope map cannot be obtained without solving the momentum conservation laws by hydraulic modeling. This can be seen clearly by a test of applying the Manning equation using bottom slope obtained from the DEM to derive a "velocity" map, which results in an extremely high velocity, which is rather unrealistic in case of a gradual flooding. The procedures are as shown below.

Assuming the velocity for each cell can be calculated using the Manning equation (Roberson et al, 1988) with the bottom slope:

$$v = \frac{1}{n} R_h^{2/3} S_f^{1/2} \qquad (5.8)$$

where,

n Roughness, Manning number

S_f Steepest slope of each cell to its neighbor cells,

R_h Hydraulic radius for each cell calculated for a rectangular cross section:

$$R_h = \frac{B*h}{2h+B} \qquad (5.9)$$

where h is inundation depth [m], and B is cell size [m]

Using the inundation map obtained with GIS-based approach, a "velocity" map is calculated for the scenario of the dike break at Elbe km 413.5 (see section 5.4.2). The difference with the velocity map obtained with SOBEK1D2D is shown in Figure 5.12.

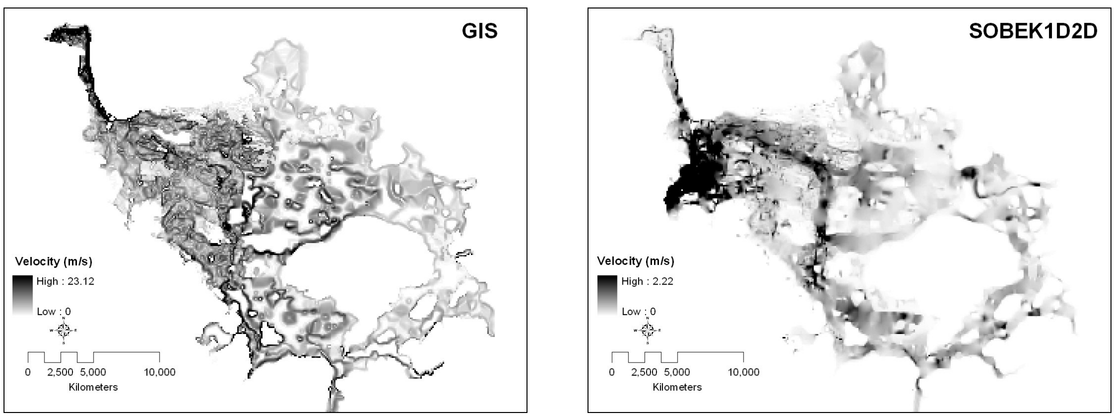

Figure 5.12 *Velocity maps obtained using GIS approximation and SOBEK1D2D*

As shown in Figure 5.12, the maximum velocity obtained with the GIS approximation is 23.12 m/s, which is much larger than the maximum velocity obtained with SOBEK1D2D (2.22 m/s). Moreover, the high velocity area is missing in the GIS-based approach map, as can be inferred from the maps.

Nevertheless, the result indicates that the GIS approximation cannot provide correct velocities without taking into account dynamics and momentum consideration, which provides the water surface slope. Thus, to obtain a more accurate velocity map, full hydrodynamic computations are essential.

5.4.5 Discussion

GIS approximation can be used for rapid FRA for inundation modeling. With the GIS approximation a key aspect of flooding, the maximum inundation depth, can be obtained with reasonable accuracy. However, this method cannot show a distribution of velocity, although the characteristics of momentum can be approximated using elevation data and land use data. The difference with the full hydrodynamics model is large. This means the GIS-based method cannot provide the dynamics of momentum causing the disaster.

5.5 Uncertainty Analysis

The comparison of results shows a significant uncertainty contribution to damage from the elevation data, which requires more attention for FRA. Thus, the propagation of uncertainties from the elevation data and roughness have been analyzed.

Due to the large computational load with SOBEK1D2D, the UA has been carried using a method based on *scenarios trees* as described in Chapter 2. Using the scenario tree method, nine scenarios are set up, consisting of different combinations of three types of a DEM (aggregated with minimum, mean and maximum neighbor elevations), with three types of roughness map (maps obtained with minimum, mean and maximum Manning *n* according to literatures). The results are shown in Table 5.7.

Table 5.7 *statistics of different scenarios result with combination of different DEM and roughness maps*

Scenario	Damage (Euro)		Area (km^2)	Spatial statistics			
				Inundation depth (m)		Maximum velocity (m/s)	
DEM_Roughness	Total million	Mean		Max	Mean	Max	Mean
min_min	124	17889	432	7.66	1.38	2.49	0.0770
min_mean	115	14165	422	7.62	1.21	2.20	0.0652
min_max	104	9348	411	7.58	1.04	1.97	0.0431
mean_min	85.2	8578	411	4.95	1.10	2.38	0.0666
mean_mean	70.3	1842	383	4.88	1.00	2.11	0.0619
mean_max	65.0	1974	348	4.83	1.25	1.87	0.0497
max_min	50.1	1556	336	4.88	1.17	2.02	0.0781
max_mean	47.9	1535	327	4.80	1.12	1.78	0.0707
max_max	47.0	1530	321	4.73	1.09	2.14	0.0474

Note: the first column item indicates the combination of DEM with roughness, for example *min_min* means map of DEM aggregated using the *minimum* value of the neighbor cell combing with map of *minimum* Manning *n*.

The total damage and inundation depth show a decreasing trend with the aggregation method changing from minimum to maximum. Such a trend doest not appear for the maximum velocity. However, if considering roughness change only, the trend is clear for each aggregation method that velocity decreases when roughens increases. For example, for the minimum aggregated DEM, the flow velocity is reduced from 2.49 m/s to 1.97 m/s using minimum and maximum roughness.

The trend can be clearer when the indicators are categorized according to their uncertainty (Table 5.8). As shown in Table 5.8, the DEM does not affect significantly the change of velocity, but only the total damage and inundation depth. A change in the roughness affects the velocity more significantly.

Table 5.8 *statistical values of scenarios categorized according to uncertainty contribution*

Scenario		Damage (Euro)		Area (km^2)	Spatial statistics			
					Inundation depth (m)		Maximum velocity (m/s)	
		Total million	Mean		Max	Mean	Max	Mean
DEM	Min	114	13801	422	7.62	1.21	2.22	0.0618
	Mean	73.5	4131	381	4.89	1.11	2.12	0.0594
	Max	48.3	1540	328	4.81	1.13	1.98	0.0654
Roughness	Min	86.4	9341	393	5.83	1.22	2.30	0.0739
	Mean	77.8	5847	377	5.77	1.11	2.03	0.0659
	Max	72.2	4284	360	5.71	1.13	1.99	0.0467

The effect is shown more clearly in Figure 5.13, which presents the change of total damage against the aggregation method for the DEM, and flow velocity against change of roughness.

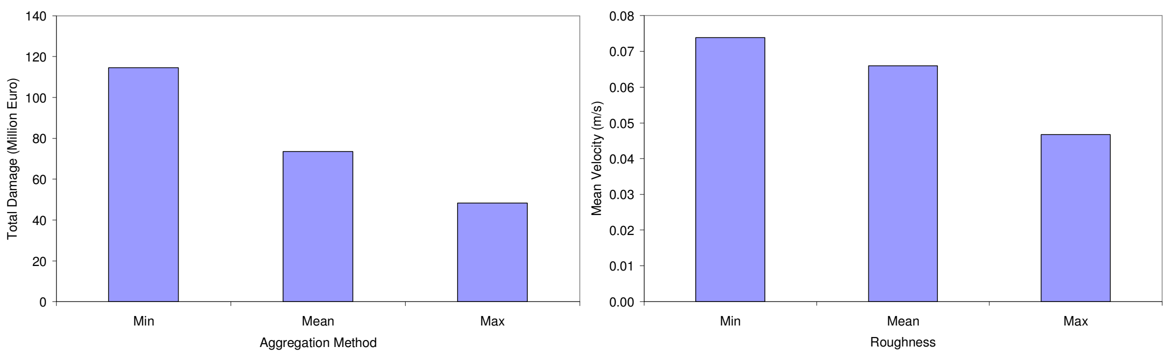

Figure 5.13 *Two most sensitive cases: (a) total damage associated with different aggregation methods to total damage, and (b) velocity with change of roughness*

Figure 5.13 clearly shows a significant uncertainty contribution to the flood damage due to the change of aggregation methods (Figure 5.13a), and a clear trend of velocity change according to the change of the roughness is also found (Figure 5.13b), that is, the higher the roughness, the lower the average velocity. The change of the flow velocity according to the change of roughness is 3-9% for the maximum velocity, and 6-24% for the average flow velocity over the mean value for each scenario. It can be also found that the change of the roughness does not affect total damage significantly although the maximum flow velocity is changed significantly. This, however, could be different in case the flood damage function would incorporate the effect of flow velocity.

This conclusion can be also found in the sensitivity analysis. Sensitivity analysis has been carried out on the two variables, DEM and roughness value, which propagates uncertainties through the hydraulic model. The results are plotted in Figure 5.14.

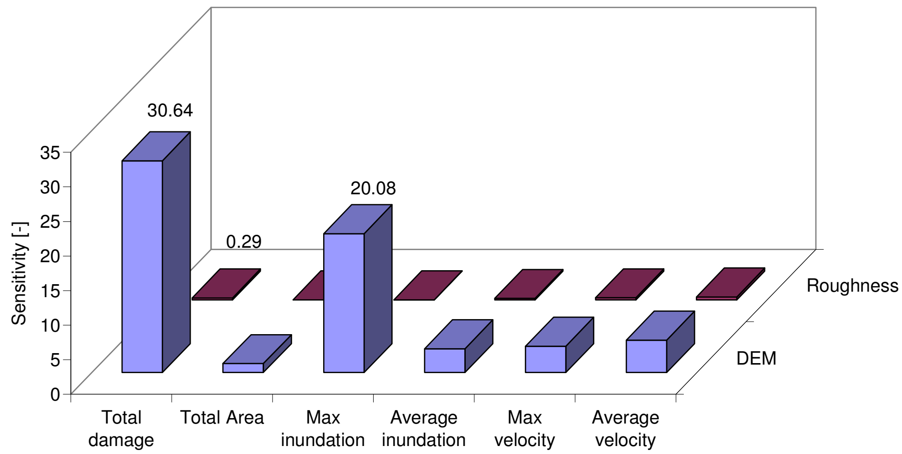

Figure 5.14 *Sensitivity of the DEM and roughness*

As shown in Figure 5.14, the sensitivity of total damage to DEM is about 30.64, sensitivity of maximum inundation to DEM is 20.08, while sensitivity of total damage to roughness is 0.29. In agreement with what has been observed previously, the results show that the elevation data contribute more to the uncertainty in the flood damage than the roughness data.

5.6 Conclusion

A comparison has been made of a full dynamic and a GIS-based approach for IFRA, with a dike break near Sandau as case study. The societal relevance lies in the use of an intentional dike break for sacrificial flooding, to protect downstream areas which are more economically vulnerable to flood damage and/or hazard to people. A dike break near Sandau can protect the town of Wittenberge. The negative, local effects of this measure consist of the damage resulting from inundation depths and flow velocities in the area that is flooded as a direct result of the dike break. Non-local effects consist of the downstream reduction of water levels and damage reduction for a particular flood event.

The simulation results obtained with SOBEK1D2D indicate that the town of Wittenberge (Elbe km 445) is best protected by breaking the dike at Elbe km 413.5 with a width of 150-200 m. The water level reduction at the gauge station of Wittenberge is around 40 cm, a significant reduction for flood protection. For a 200-year flood, the local damage near Sandau is around 130 million euro. In order to estimate the damage reduction downstream a dike break upstream of Wittenberge was simulated as well, with 420 million euro damage as result. This points to a net benefit of around 300 million euro.

For rapid FRA, a GIS approximation is more desirable for inundation modeling in view of the computing time, as well as less cost such as model calibration/validation and data demands. However, because of the assumption that the water level is horizontal in the inundated area, such approach can only be applied for a relatively small area. An approximation has been made using GIS-based approach. The resulting flood damage and to a somewhat lesser extent the inundation depth is in reasonable agreement with those obtained with the SOBEK1D2D model. However, it has been found difficult to approximate velocity without take into account changes in momentum, which implies that the water surface slope cannot be obtained (so far) by using GIS approximation only.

In the UA it is found that the flood damage computation is very sensitive to the propagation of uncertainty in the elevation data. The aggregation method of the DEM contributes most uncertainty to the damage computation when only inundation depth is taken into account. Therefore, the aggregation of DEM, if necessary, must be handled with care.

The UA also shows that the flow velocity is sensitive to the roughness values in the hydrodynamic modeling. Therefore, it is advised to obtain an accurate relationship between land use and roughness value (for example Manning number *n*) which is essential when flow velocity is taken into account for FRA.

Chapter 6

Conclusions and Recommendations

This chapter summarizes the main content of the thesis, addressing issues related to the design and application of Decision Support Systems (DSS) for Integrated River Basin Management (IRBM) with particular emphasis on Flood Risk Assessment (FRA). The concept of appropriate modeling is followed to derive a systematical framework for the design of a DSS for Integrated Flood Risk Assessment (IFRA). A sound basis is established for selecting component models as well as for establishing criteria for system evaluation, accounting for effects due to uncertainty propagation. The significance of including effects of flow velocity when assessing flood damage is demonstrated using the risk matrix approach. Results were obtained for a case study on the Elbe River in Germany, applying a fully integrated 1D2D hydrodynamic approach. For rapid FRA at reduced computing time, a GIS-based tool was developed which can provide a first indication of flood risk, including potential effects of flow velocities on flood damage. Answers to the research questions formulated in Chapter 1 are provided, together with recommendations for future research aimed at achieving more effective IFRA.

6.1 Synopsis

6.1.1 Issues in IRBM and DSS development

There is evidence of a growing gap between scientific knowledge available and practical needs required for IRBM (Westcoat, 1992; Nienhuis et al., 1998; Jonker, 2002; Jeffrey and Gearey, 2003). This gap becomes manifest when e.g. management objectives or operational requirements are not clearly defined and correctly translated into the different “languages” that are used by scientists and end users; or when the development of measures is not based on adequate scientific principles, resulting in ineffective implementation of the proposed measures. Moreover, the proper choice of the various temporal and spatial scales and the specific questions at different management levels increase the complexity of IRBM. The problems related to IRBM call for accurate and efficient tools to support decision making in relation to management activities. Therefore, the challenge is to design tools that are appropriate for describing changes in the objective variables. Preferably, such instruments should not be excessively complex nor overly simple, but appropriate to analyze the problems encountered in IRBM.

6.1.1.1 Model selection

Clearly, IRBM can benefit from an integrated DSS. However, an integrated DSS remains difficult to design due to the issue of how to select models that are neither excessively complex nor overly coarse for their intended purpose. In addition, a systematic evaluation approach providing quantitative indicators for DSS performance seems to be lacking. Moreover, uncertainty has been found to play a significant role in both modeling activities and in management practices, but is still hardly accounted for in DSS development. This may result in inappropriate tools to assess proposed measures for IRBM. Therefore, improvements are needed to develop a DSS of appropriate complexity taking into account effects of uncertainty.

A system analysis approach is often followed in the design of most DSSs (Loucks, 1995; Jamieson and Fedra, 1996a; Schielen and Gijsbers, 2003; Fassio et al, 2004; Mysiak et al., 2005). This implies a clear design logic with common steps of problem definition, management alternative identification, future context prediction, model and system identification, with the ultimate objective of ranking and comparing alternatives for decision making.

Based on the system analysis approach, two types of DSS design architectures can be discerned, often referred to as *user-oriented* and *knowledge-driven*. The design is either aimed at developing a DSS serving a specific problem for a specific river basin, or a generic tool that can be used in any river basin to deal with a range of problems. The difference in design philosophy also leads to different ways of model selection: the user-oriented approach tends to make use of readily available models and data, whereas the knowledge-driven approach often aims to incorporate more elaborate mathematical models to be prepared for possible additional requirements that may occur in the future. In general a user-oriented approach puts emphasis on the participation of end users which may lead to an ill-structured design if insufficient disciplinary knowledge is involved, whereas a knowledge-based approach is likely to describe the physical system more closely, but might lead to an overly complex system, when too many details are included in the system which might not be needed to analyze the problem at hand. This may well result in unnecessary cost in terms of data demands or work load.

However, in the design of DSSs, the question HOW models should be selected with a minimum but adequate complexity has not received adequate attention. The selection of models to be included in an integrated DSS is not possible without clear criteria or a systematic procedure. Although a comparative approach for model selection has been widely used (Grijspeerdt et al., 1995; Jamieson and Fedra, 1996a; Venterink and Wassen, 1997; Wood et al., 1998; Schielen and Gijsbers, 2003; Mysiak et al., 2005), this is only effective when a model can be selected from various already available candidates. The dependence of model performance on the experience of the particular modeler, further increases the difficulty of model selection.

6.1.1.2 Uncertainty analysis

Once a DSS has been set up, the question arises how to quantify its overall performance, or in other words, how to evaluate the reliability and uncertainty associated with the answers provided. Some attempts have been made to evaluate the quality of a DSS (Reitsma, 1996; Finlay and Wilson, 1997; Poon and Wagner, 2001). According to Potts et al. (2001), the generic problem of DSS evaluation is the lacking definition of quality and of methods to assess this quality. Finlay and Wilson (1997) refer to about 50 overlapping, ambiguous validity concepts for measuring quality in the literature which differ in their chosen approach and in the features of the DSS addressed. Despite these ambiguities in validity concepts developed so far, some of the success factors, also known as critical success factors (Poon and Wagner, 2001), are commonly agreed upon: UA is invariably found to be important for evaluating model performance (Snowling and Kramer, 2000), as well as for distinguishing the impact of alternative measures (De Kok and Wind, 2003).

However, UA is often applied to individual (component) models only (e.g. Crosetto et al., 2001; Hanna et al., 2001). The propagation of model uncertainty through an integrated system involving multiple different component models has rarely been reported. This triggered the need to study uncertainty and its propagation through an integrated DSS, as carried out in this thesis.

In summary, to bridge the gap between scientific principles and practical needs in IRBM, a systematic design approach is to be followed and the issues of model selection and system evaluation should be explicitly addressed, so that a DSS is developed which is not excessively complex but does contain appropriately sufficient knowledge.

6.1.2 A framework for DSS design

Based on the knowledge and experience obtained during the development of a pilot DSS for the Elbe River in Germany, the concept of appropriate modeling has been elaborated in this thesis with regard to model selection and system performance evaluation. In order to achieve the objectives a DSS, the concept of appropriate modeling can be invoked to arrive at a framework for the design. This framework should be based on an iterative and interactive design process consisting of two phases of analysis: the qualitative analysis phase and the quantitative analysis phase, aiming to determine appropriate model and system complexity which can facilitate decision making.

The qualitative analysis phase involves: problem definition, study of physical conditions including data availability, followed by identifying the causal relationships connecting relevant processes and variables to the objectives and measures. The result of this phase is a conceptual system network. The quantitative analysis phase focuses on the detailed formulation of models including set up (selection of a particular model), calibration and validation, as well as on outcome variation obtained from UA.

The concepts of internal consistency (De Kok and Wind, 2002) and environmental indicators (EEA, 1999) are particularly useful in appropriate modeling related to IRBM. These two concepts aim at reducing the complexity of the DSS by detailed analysis of the process of model and variable selection based on causal analysis. In order to apply these two concepts, a double-direction search method was elaborated in this thesis. This method determines the complexity of models and modeling systems within the two boundaries provided by the physical environment and the environmental indicators. This differs from the more conventional (inter)comparative approach applied to model selection.

In addition to the technical analysis, during the development of the IFRA, *communication* between modelers and end users has been proved beneficial. Through intensive discussion, end user’s confidence of the DSS was increased because of the progressive improvement of their knowledge of the processes. By knowing what has been put into the system and how the system produces outcomes, the end users obtained more insight into the DSS, which improved the effectiveness of the DSS and its applicability.

6.1.3 Flood risk assessment

Difficulties encountered in the design of a DSS can be found in many applications of integrated FRA. Typical problems for the development of a FRA system are: what risk assessment approach should be used to serve which management objectives? What hydraulic models should be used for which risk model? How to evaluate and estimate the effectiveness of a proposed flood management measure? How to include effects of flow velocity in FRA? How to obtain a rapid FRA? The following sections briefly review these questions and the way they were dealt with in this thesis.

6.1.3.1 Issues in FRA

Flood risk assessment used to be based on a statistical risk-analysis approach (CUR, 1990; Stedinger, 1997; Vrijling et al., 1998). In the statistical approach (e.g. Vose, 1996), flood risk is defined as the product of the probability of flood occurrence, and its consequent damage. This approach is particularly useful to support long-term planning such as the implications of dike construction (CUR, 1990) and is usually applied at a larger spatial scale (size of several hundred kilometers). More recent risk assessment studies focus on detailed simulations using a physically-based approach, where the flood damage is calculated for specific flood events using numerical inundation models (Horritt and Bates, 2002; Van der Sande et al., 2003). This approach provides insight into the damage associated with a particular flood event, which can be used for short-term planning such as the operation of a retention basin during the flooding period. Integrated FRA should allow both for long-term planning and for short-term operation pertaining to different spatial scales, and should therefore include both risk assessment approaches.

Experiences with flood events (like the recent Cornwall flood event or the Asian Tsunami) indicate that flow velocity can play a significant role in causing flood damage. However, so far this is not quantitatively expressed or systematically accounted for in classical risk assessment methods. Including the effect of flow velocity should be of important concern in the design of IFRA, as elaborated in this thesis.

In order to measure DSS performance, an effective evaluation method for IFRA is also needed. Studies have been made addressing UA for IFRA models (e.g. De Blois and Wind, 1996; Al-Futaisi and Stedinger, 1999; NRC, 2000; De Roo et al., 2003). However, uncertainties involved in hydraulic model structures were not taken into account, which limited the validity of the conclusions on flood damage.

6.1.3.2 A framework for IFRA

Following the concept of appropriate modeling developed within the framework of DSS design, a qualitative framework for IFRA was set up using the Elbe River DSS as case study. The framework is based on causal reasoning to explore the relationships between processes and indicators involving hydraulic as well as socio-economical parameters. Two risk approaches, namely the statistical approach and the physically-based approach, are included.

In accordance with the requirement and properties of the risk models, two types of hydraulic models are identified, namely a 1D steady flow model for constructing the boundary conditions (stage-discharge curves), and an overland flow model which can actually simulate the overland flow hydrodynamics involving two-dimensional (horizontal) flow velocities. To estimate the consequence of particular measures, UA is applied to provide the variability associated with the calculated risk/damage as the final outcome of IFRA. In order to include the additional effects resulting from flow velocity, a risk matrix method has been adapted in this thesis.

6.1.3.3 Case study 1 – Elbe_DSS

In this case study, the conceptual IFRA framework (Chapter 3) is built around risk assessment approaches, namely the statistical approach and the physically-based approach. These risk approaches are combined with their associated hydraulic models, namely the steady-state model HEC6 (HEC6 User's Manual, 1993) for the statistical approach, and the floodplain hydrodynamic model SOBEK1D2D (Stelling and Duinmeijer, 2003) for the physically-based simulation approach. To account for the contribution of flow velocity to the damage, a table combining conventional damage and effects of flow velocity, viz. the so-called risk matrix (Table 3.2), was constructed.

The area near the town of Sandau in Germany along the Elbe River is selected as the modeling area. The IFRA is tested by assessing the effectiveness of dike presence with four scenarios concerning one measure: removing the dike – with (1) and without (2) a dike at the modeling area; and breaking the dike – cases with (3) and without (4) an intentional artificial dike break. Both the statistical approach and the physically-based FRA procedure are tested for the study area. Monte Carlo Simulation (MCS) is applied to propagate uncertainties through the system for each combination of scenarios and for each FRA approach.

As a result, an operational IFRA methodology is developed which can be applied to assess the consequences of implementing flood management measures to aid in the decision making. Results of scenarios show effective risk assessment using the IFRA approach. Clear differences in the spatial distribution of risk are observed when the effect of flow velocity is taken into account. The role of uncertainty on quantifying the effect on flood risk of two measures is investigated.

6.1.3.4 Case study 2 – Risk reduction

Another case study was carried out on risk reduction along the Elbe River for flood mitigation using the deliberate dike break measure (Chapter 5). IFRA is applied to establish the benefits of intentional dike break at an upstream area, which is of lower economic importance. The results point out that this can be an effective mitigation of risk/damage.

The results also show that when the practical implications of the flood event is taken into consideration, i.e., where and how to break the dike in order to obtain an optimal downstream damage reduction - a fully 2D dynamic model is needed. This has been demonstrated by the proper selection of dike break time and dike break width, that result in a maximum reduction of the water level downstream. Such a dynamic approach is essential for short-term operational management such as evacuation, when timing is an important parameter to consider.

However, fully 2D hydrodynamic simulations can be computation time-demanding which may prevent their practical use in physically-based FRA. A more efficient inundation model would then be an alterative. Although parallel computing algorithms could be used to speed up computational performance (Mynett, 2004c), or hydroinformatics techniques like Artificial Neural Networks could be employed to emulate pre-computed scenario's (Mynett et al., 2004b), this is not yet common practice in current FRA approaches. Alternatively, an approximate method using 1D flood routing for the river section with 2D GIS technology for the floodplain area (Chapter 5), has been developed within this thesis research, referred to as the GIS-based approach.

The GIS approximation was applied for quick risk assessment at the studied area. The results show a reasonably good agreement with SOBEK1D2D simulation in terms of inundation depth and flood damage, but not for flow velocity because of the missing momentum considerations in GIS. Still, this approach approximates inundation depth and flow velocities using the concept of hydrological basins and a slope map determined from a digital elevation map.

Uncertainty analyses were carried out for the most sensitive data elements, viz. digital elevation (determining the magnitude of the gravitational driving force), and land use (determining the resistance due to roughness in hydraulic models). For DEM, the effect of the level of aggregation of cell size has been explored; uncertainty in roughness was obtained by varying the roughness associated with each land use type. The results show that uncertainty is dominated by the level of aggregation of DEM, and very little by roughness. However, uncertainty due to roughness does lead to significant changes in flow velocities, which can have a significant effect, since velocity can cause important damage contributions (like in the recent Cornwall and Asian Tsunami floods).

6.2 Conclusions

An IFRA framework has been developed to address the problem of selecting hydraulic models and to improve the overall DSS performance. The framework is founded on the principle of appropriate modeling, which aims for an appropriate level of model complexity, i.e. models that are neither excessively complex nor overly coarse. Flood management objectives can be categorized according to their long-term planning or short-term purpose. Two complementary risk approaches are available within this framework. The statistical risk-analysis based approach provides the expected annual damage as outcome. It can support long-term planning of IFRA by providing the spatial distribution of flood risk. The physically-based risk assessment approach accounts for the effects of measures or for the failure of a flood defense system. It can be used to support decision-making related to short-term flood management, including a priori activities such as risk mitigation and evacuation, as well as a posteriori activities such as damage assessment for repair after a flood has occurred.

According to the proposed IFRA framework, the choice for a particular hydraulic model should depend on the purpose of the assessment. A double-direction searching algorithm is introduced to identify the appropriate level of complexity for the hydraulic model, which can serve as model selection guideline. For the statistical approach, the key model component is the stage ~ discharges relationship, which can be determined with a relatively simple steady hydraulic flow model. For short-term flood management, hydrodynamic modeling is needed. For the river channel, a 1D hydraulic model can be applied in general. However, it is essential to apply 2D modeling for the area at risk behind the dikes, because flow velocities can contribute significantly to the flood damage.

Case studies using the developed IFRA framework have been carried out for an area near the town of Sandau along Elbe River. Based on the study, the following conclusions can be drawn, presented below in the form of answers to the research questions posed in Chapter 1.

6.2.1 How can the effect of flow velocity on flood risk be incorporated in IFRA?

Flow velocity can contribute significantly to flood damage. It can be incorporated into FRA using the risk-matrix approach by combining conventional inundation damage with effects due to flow velocity. The results for the Elbe case study show a significant difference in the spatial distribution of the risk when flow velocities are taken into account. In this case study, the risk levels were defined in consultation with end-users.

Strictly speaking, the risk matrix concept employed in this thesis is more suitable for a statistical risk approach than for a physically-based approach because of the underlying statistical concept. The velocity index used in the risk matrix is the normalized spatial velocity rather than the real velocity value (Chapter 3.4.3). It is used to reflect the momentum characteristics of the river basin and is not associated with a specific flooding event, which could well provide a different spatial velocity distribution map according to instantaneous flooding conditions such as overtopping area

or dike break parameters. Therefore, careful interpretation is required when using the risk matrix concept in a physically-based approach.
In any case, proper fully 2D hydrodynamic modeling is essential to assess the spatial distribution of the maximum flow velocity corresponding to the geography and land use of the area.

6.2.2 How can uncertainty analysis be applied to IFRA?

Uncertainty analysis can support IFRA by presenting comprehensive risks indicators. UA can also quantify uncertainty contributions from each component model and of the data used for calibration and validation of the various model components. The UA results also indicate the contribution from the various data and model sources to the overall uncertainty in the outcome of the IFRA.

The uncertainty analyses carried out for FRA in the Sandau case study point to a significant uncertainty contribution from the aggregation method for the elevation data (DEM). The aggregation method introduces large uncertainty in the computation of the flood damage, which is due to uncertainty in the inundation depth computation using the processed DEM.

For large computational loads involving long timer computations, a simplified Monte Carlo simulation - the scenario tree method - is proposed and applied in this research. This method propagates uncertainties through hydraulic models to risk models without requiring large runs of the 2D hydraulic model. The results show distinguishable uncertainty distributions, which also indicates the effectiveness of the UA approach.

6.2.3 Can GIS technology provide a useful approximation for rapid IFRA?

In general a fully 2D hydrodynamic model, such as the SOBEK1D2D system used in this thesis, should be applied to obtain time varying inundation depth and velocity distribution. However, to avoid time consuming computations involved in complex hydrodynamic modeling, a GIS approximation could be used to obtain a first indication of inundation depth and flow velocity to be used for rapid FRA.

For rapid assessment, indications of local water level are sufficient, and a GIS-based approach could well be appropriate. By applying a stepwise search algorithm using storage functions for each sub-basin, the inundation depth and area can easily be determined. Based on a mass balance approach, assuming water level is horizontal in the inundated area, the results obtained with the GIS approximation for the Elbe case study agree quite well in terms of inundation depth and flooded area.

A drawback of this approach is that it does not contain the underlying processes or dynamics when flow velocity needs to be taken into account for, which could result in considerably different damage assessment. GIS technology can be used to obtain approximate velocity indications if water surface slope can be approximated in some way. However, this problem cannot be resolved without carrying out full hydrodynamic computations.

6.3 Recommendations

Based on the lessons learned in this research, the following recommendations are provided for future elaboration:

6.3.1 The role of scientific principles in DSS design for IRBM should be enhanced

By following the concept of appropriate modeling in the process of designing a DSS for IRBM, the models applied are neither excessively complex nor overly coarse, while satisfying the functionality requirements of all end-users. In order to achieve this, the role of scientific principles should be enhanced. Thus, attention should be given to end-users as well as modelers, identifying management problems c.q. objectives, as well as the state-of-the-art in knowledge and its implementation. These aspects determine the appropriate complexity of the DSS, as well as the adequate representation of the physics. Communication between modelers and end-users makes the process of appropriate modeling interactive and iterative.

6.3.2 Quantitative relations between flow velocity and flood damage should be established

Although the effect of flow velocity can be incorporated using the risk-matrix approach applied in this research, a clear and direct relationship between flow velocity and resulting damage remains lacking. Thus, future work should focus on establishing the quantitative relationship between damage and velocity. Historical measurements of flood loss at flooding areas could contribute to establishing such a relationship. Quantitative inclusion of other important variables such as flood duration should also be considered.

6.3.3 GIS technology can provide approximation of hydraulic characteristics

In hydraulics, the flow velocity is largely determined by the slope (gravity component driving the flow) and the roughness of the flow area (resistance component opposing the flow), so an indication for the velocity characteristics could be obtained using slope and roughness of the flooding area, which is information that is readily available from GIS systems that are becoming more widely available in practice.

6.3.4 A sound scientific standard for flood damage functions is needed.

Applying commonly used flood damage functions based on inundation depth ~ percentage damage curves, shows large deviations from results obtained from physically-based hydrodynamic simulation, leading to significant uncertainty in risk assessment. Therefore, developing depth ~ damage (%) curves based on sound scientific standards will considerably improve the risk assessment To obtain such curves, a standard land uses classification, such as the one used in CORINE land cover classification, is also important.

6.3.5 Scenario tree method could reduce the computational load in uncertainty analysis

It has been found that uncertainty analysis is of great benefit to both the development and the implementation of IFRA. However, due to the large computation loads involved in IFRA, conventional UA methods such as Monte Carlo Simulation might be impractical. This could be overcome by using a scenario tree method as applied in this research. This method can be quite useful to perform uncertainty analyses on integrated systems, particularly when large computations are needed, such as in case of two or three dimensional hydrodynamic modeling.

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List of Abbreviations

AM	Appropriate Modeling
DEM	Digital Elevation Model
DKKV	Deutsches Komitee für Katastrophenvorsorge e. V.
DSS	Decision Support System
EAD	Expected Annual Damage
EEA	European Environmental Agency
GIS	Geographic Information System
FDBR	Flow Dam Break Reach
FRA	Flood Risk Assessment
GIS	Geographical Information System
IDSS	Integrated Decision Support System
IFRA	Integrated Flood Risk Assessment
IKSE	Internationale Komission zur Schutz der Elbe
IKSR	Internationale Kommission Zum Schutz des Rheines
IRBA	Integrated River Basin Management
LHS	Latin Hypercube Sampling
OAT	One-at-a-time design
MCS	Monte Carlo Simulation
RBM	River Basin Management
SA	Sensitivity Analysis
UA	Uncertainty Analysis

List of Symbols

a_g, b_g	Coefficient of Gumbel distribution (Eq. 3.4)
a_p, b_p	Coefficients of fitted power function for discharge ~ duration [-] (Eq. 3.7)
a_r, b_r	Coefficients of rating curves [-] (Eq. 3.3)
A_j	Area of computing cell j [m^2] (Eq. 5.6)
B	Computation cell size [m] (Eq. 5.9)
B_0	Initial width of the breach [m] (Eq. V.9)
$B(t)$	Width of the breach at point in time t [m] (Eq. V.9)
c	The deviation for the unit normal enclosing probability α (Eq. 2.8)
C	Chézy coefficient [$m^{1/2}/s$] (Eq. V.1)
C	Consequence in risk analysis, e.g. the damage, loss of life (Eq.2.1)
C_e	Expansion loss coefficient [-] (Eq. IV.1)
d	Depth below plane of reference [m] (Eq. V.1)
$d_i(X)$	i^{th} factor at a given point X (Eq. 2.5)
D	Flood duration [day] (Eq. 3.7)
D_i	Flooding duration [day] (Eq. 3.6)
D_h	Diffusion coefficient [km^2/hr] (Eq. 5.5)
Dq_j	The original flood duration from historical data [day] (Eq. 3.6)
$\langle D \rangle_{ij}$	Expectation value of percentage flood damage at computing cell (i,j) [Euro] (Eq. 3.2)
f_1	Factor1, constant factor (input parameter) [-] (Eq. V.9)
f_2	Factor 2, constant factor (input parameter) [-] (Eq. V.9)
$fd(h(q))$	Flood damage function, as a function of inundation depth [%] (Eq. 3.2)
f_{damage}	Flood damage function (Eq. 3.1)
$f(q)$	Probability density function for the Gumbel distribution (Eq. 3.2)
g	Gravity acceleration [m/s^2] (Eq. IV.1)
h	Inundation depth [m]
h_e	Energy head loss [m] (Eq. IV.1)
h_{up}, h_{down}	Upstream and downstream water level at point-in-time t [m] (Eq. V.9)
$h_{ij}(q)$	Inundation depth in cell (i,j) [m] (Eq. 3.2)
$H_{additional}$	Safe height [m] (Eq. 4.2)
H_d	Dike height [m] (Eq. 4.1)
H_f	Freeboard [m] (Eq. 4.1)
Hi	Water level [m] (Eq. 5.6)
Hr	Water level associates with certain return period [m] (Eq. 4.2)
H_{wave}	Wave height [m] (Eq. 4.2)
H_{wind}	Wind height [m] (Eq. 4.2)
K	Total conveyance
L	Length of the modeled river section [km] (Eq. 5.5)
m	Sample size for Monte Carlo simulation (Eq. 2.8)
m_e	Number of effective flood events [-] (Eq. 3.8)
n	Roughness, Manning number (Eq. 5.8)
ne	Number of total amount of flood event [-] (Eq. 3.8)
P	Probability of failure [%] (Eq. 2.1)

$p_{failure}$	Probability of failure (Eq. 3.1)
q^*_{ij}	Critical discharge, calculated using rating curve at computing cell (i,j) [m^3/s] (Eq. 3.3)
$q(t)$	Discharge [m^3/s] (Eq. 5.5)
Q_i	Peak discharge value associated with flood duration of D_j [m^3/s] (Eq. 3.6)
Q_p	Peak value of discharge [m^3/s] (Eq. 3.7)
Q_T	Peak discharge associated with return period T [year] [m^3/s] (Eq. 3.5)
R	Risk (Eq. 2.1)
R_h	Hydraulic radius [m] (Eq. 5.8)
S	Sensitivity [%] (Eq. 2.2)
S_f	Steepest slope of each cell to its neighbor cells [-] (Eq. 5.8)
s^2	Variance (Eq. 2.8)
t	Time [hr] (Eq. 5.5; Eq. V.1)
t_0	The point-in-time when the maximum breach-depth (z_{min}) is reached [min] (Eq. V.9)
t_{start}	Point-in-time at which the breach starts to develop [min] (Eq. V.9)
T	Return period [year] (Eq. 3.5)
T_0	Time span over which the breach having a constant initial width (B_0) is lowered from its initial crest level (z_{crest}-*level*) to its final crest level (z_{min}) [hr] (Eq. V.9)
u	Translation coefficient [km/hr] (Eq. 5.5); velocity in x-direction [m/s] (Eq. V.1)
u_c	Constant critical flow velocity sediment/soil (input parameters) [m/s] (Eq. V.9)
v	Velocity [m/s] (Eq. 5.8); velocity in y-direction [m/s] (Eq. V.7,ab)
$v^N_{ij,k}$	Normalized velocity at each cell ij for flood event k [-] (Eq. 3.8)
v_{ij}	Flow velocity at cell (ij) [m/s] (Eq. 3.9)
$v^s_{\max}$	Spatial maximum flow velocity [m/s] (Eq. 3.9)
V_i	Volume associated with water level H_i [m*3*] (Eq. 5.6)
V_1,V_2	Mean velocities (total discharge/total flow areas) at ends of reach [m/s] (Eq. IV.1)
WS_1,WS_2	Water level at ends of the modeled river stretch [m] (Eq. IV.1)
x	Distance [m] (Eq. V.1)
x_i	Model input variable (Eq.2.2)
X	Vector of input variables (Eq. 2.5)
X^*	'Base' value of a vector of input variables (Eq. 2.5)
$X^{(2)}$	The 2nd vector of input variables (Eq. 2.6)
y	Model output variable (Eq. 2.2)
$z_{crest\text{-}level}$	Elevation of the crest-level of the dike [m] (Eq. V.9)
z_j,	Elevation [m] (Eq. 5.6)
z_{ij}	Elevation at computing node (i,j) [m] (Eq. 3.3)
z_{dikei}	Dike height at row i [m] (Eq. 3.3)
z_{min}	Elevation of the bottom after dike-breach [m] (Eq. V.9)
$z(t)$	Elevation of the dike-breach at point-in-time t [m] (Eq. V.9)
σ	Standard deviation (Eq.2.3)
Δk_j	The variation of input variable x_j (Eq. 2.4)
Δ	Predetermined multiple of $1/(p-1)$, (Eq. 2.5)
Δt	Time step [s] (Eq. V.2)
Δx	Space step [m] (Eq. V.2)
α	Confidence interval [-] (Eq. 2.8)

α_1, α_2	Velocity coefficient for flow at ends of reach [-] (Eq. IV.1)
ϖ	Unit wide [-] (Eq. 2.8)
$\widehat{E}(Y)$	Expected value of output variable (Eq. 2.9)
$\widehat{V}(Y)$	Variance of the output variables (Eq. 2.9)
ζ	Water level above plane of reference [m] (Eq. V.1)

Appendix I Reclassification of CORINE Land Use Data

Flood damage runction		Roughness		CORINE	
Code	Class	Code	Class	Code	Class
1	Urban Area	1	Urban Area	1	'CONTINIOUS URBAN FABRIC'
1	Urban Area	1	Urban Area	2	'DISCONTINUOUS URBAN FABRIC'
2	Industry	2	Industry	3	'INDUSTRIAL OR COMMERCIAL UNTS'
3	Traffic	3	Traffic	4	'ROAD AND RAILNETWORKS AND ASSOCIATED LAND'
3	Traffic	3	Traffic	5	'PORT AREAS'
3	Traffic	3	Traffic	6	'AIRPORTS'
6	Others	4	Construction site	7	'MINERAL EXTRACTION SITES'
6	Others	4	Construction site	8	'DUMP SITES'
6	Others	4	Construction site	9	'CONSTRUCTION SITES'
6	Others	5	Artificial, non-agricultural vegetated areas	10	'GREEN URBAN AREAS'
6	Others	5	Artificial, non-agricultural vegetated areas	11	'SPORT AND LEISURE FACILITIES'
4	Agriculture	6	Agriculture	12	'NON-IRRIGATED ARABLE LAND'
4	Agriculture	6	Agriculture	13	'PERMANENTLY IRRIGATED LAND'
4	Agriculture	6	Agriculture	14	'RICE FIELDS'
4	Agriculture	6	Agriculture	15	'VINEYARDS'
4	Agriculture	6	Agriculture	16	'FRUIT TREES AND BERRY PLANTATIONS'
4	Agriculture	6	Agriculture	17	'OLIVE GROWES'
4	Agriculture	6	Agriculture	18	'PASTURES'
4	Agriculture	6	Agriculture	19	'ANNUAL CROPS ASSOCIATED WITH PERMANENT CROPS'
4	Agriculture	6	Agriculture	20	'COMPLEX CULTIVATION PATTERNS'
4	Agriculture	6	Agriculture	21	'LAND PRINCIPALLY OCCUPIED BY AGRICULTURE
4	Agriculture	6	Agriculture	22	'AGRO-FORESTRY AREAS'
5	Forest	7	Forest	23	'BROAD-LEAVED FOREST'
5	Forest	7	Forest	24	'CONIFEROUS FOREST'
5	Forest	7	Forest	25	'MIXED FOREST'
5	Forest	7	Forest	26	'NATURAL GRASSLAND'
5	Forest	7	Forest	27	'MOORS AND HEATHLAND'
5	Forest	7	Forest	28	'SCLEROPHYLLOUS VEGETATION'

5	Forest	7	Forest	29	'TRANSITIONAZL WOODLAND-SHRUB'
6	Others	8	Vegetation free	30	'BEACHES
6	Others	8	Vegetation free	31	'BARE ROCK'
6	Others	8	Vegetation free	33	'BURNT AREAS'
6	Others	8	Vegetation free	34	'GLACIERS AND PERPETUAL SNOW'
6	Others	9	Wetland	35	'INLAND MARSHES'
6	Others	9	Wetland	36	'PEATBOGS'
6	Others	9	Wetland	37	'SALT-MARSHES'
6	Others	9	Wetland	38	'SALINES'
6	Others	9	Wetland	39	'INTERTIDAL FLATS'
7	water bodies	10	water bodies	40	'WATER COURSES'
7	water bodies	10	water bodies	41	'WATER BODIES'
7	water bodies	10	water bodies	42	'COASTAL LAGOONS'
7	water bodies	10	water bodies	43	'ESTUARIES'
7	water bodies	10	water bodies	44	'SEA AND OCEAN'
-9999	NODATA	-9999	NODATA	49	''
-9999		-9999	'950'	50	''
-9999		-9999	'951'	51	''
-9999		-9999	'952'	52	''

Note: Data source (EEA, 2002).

Appendix II Table of Roughness: Manning *n* ~ Land Use

Code	n_normal	n_max	n_min	Landuse legend (Roughness reference)
1	0.2	0.2326	0.1674	CONTINIOUS URBAN FABRIC –residential building (Van der Sande et al., 2003).
2	0.015	0.0174	0.0126	DISCONTINUOUS URBAN FABRIC -Assumption.
3	0.2	0.2326	0.1674	INDUSTRIAL OR COMMERCIAL UNTS –industrial company/agency (Chow, 1959; Van der Sande et al., 2003).
4	0.015	0.0204	0.0132	ROAD AND RAILNETWORKS AND ASSOCIATED LAND – road has a roughness of 0.013 and railroad 0.033 (Van der Sande et al., 2003). The assumption is made that 90% is road.
5	0.0536	0.0623	0.0449	PORT AREAS - concrete is 0.017 and building is 0.200 (Chow, 1959; Van der Sande et al., 2003). Assume that 80% is concrete and 20% is building.
6	0.0536	0.0623	0.0449	AIRPORTS -Same as port area.
7	0.12	0.1396	0.1004	MINERAL EXTRACTION SITES – sand deposit area (De Roo, 1999; Van der Sande et al, 2003).
8	0.12	0.1396	0.1004	DUMP SITES –sand deposit area ((De Roo, 1999; Van der Sande et al. 2003).
9	0.12	0.1396	0.1004	CONSTRUCTION SITES - sand deposit area (De Roo, 1999).
10	0.1	0.1163	0.0837	GREEN URBAN AREAS - private/public garden (Chow, 1959).
11	0.0725	0.0843	0.0607	SPORT AND LEISURE FACILITIES - building is 0.200 (Van der Sande et al., 2003), and short grass is 0.030 (Chow, 1959). Assume that 75% is grass.
12	0.035	0.04	0.03	NON-IRRIGATED ARABLE LAND - pasture, high grass (Chow, 1959).
13	0.035	0.03	0.025	PERMANENTLY IRRIGATED LAND - pasture, short grass (Chow, 1959).
14	0.035	0.0349	0.025	RICE FIELDS - short grass, water (Chow, 1959; Van der Sande et al., 2003).
15	0.1	0.1163	0.0837	VINEYARDS - medium to dense brush, in summer (Chow, 1959).
16	0.15	0.2	0.1	FRUIT TREES AND BERRY PLANTATIONS -Average between 0.200 for trees (Van der Sande et al., 2003), 0.100 (Chow, 1959) and 0.150 (www.Imnoeng.com).
17	0.15	0.2	0.1	OLIVE GROWES
18	0.035	0.4071	0.0293	PASTURES - IVR-DSS grants a roughness of 0.033 (Chow, 1959; www.Imnoeng.com); 0.259 (Van der Sande et al., 2003).
19	0.035	0.4071	0.0293	ANNUAL CROPS ASSOCIATED WITH PERMANENT CROPS - mature row crops (Chow, 1959).
20	0.05	0.0582	0.0419	COMPLEX CULTIVATION PATTERNS - maximum value of mature field crops (Chow, 1959).
21	0.05	0.0582	0.0419	LAND PRINCIPALLY OCCUPIED BY AGRICULTURE, WITH SIGNIFICANT AREAS OF NATURAL VEGETATION - scattered brush, heavy weeds (Chow, 1959).
22	0.12	0.1396	0.1004	AGRO-FORESTRY AREAS - heavy stand of timber, a few down trees, little undergrowth (Chow, 1959).
23	0.12	0.1396	0.1004	BROAD-LEAVED FOREST - heavy stand of timber, a few down trees, little undergrowth (Chow, 1959).
24	0.2	0.25	0.15	CONIFEROUS FOREST - dense willows, summer straight. Sander et al and match with roughness of 0.200 and 0.15, respectively. IVR-DSS assigns a value of 0.058 (Chow, 1959; www.Imnoeng.com).

25	0.2	0.25	0.15	MIXED FOREST - dense willows, summer straight. 0.200 (Van der Sande et al., 2003) and 0.15 (Chow, 1959).
26	0.033	0.035	0.03	NATURAL GRASSLAND -IVR-DSS, grass. Chow (1959) gives a medium value for short grass of 0.030 and 0.035 for high grass.
27	0.033	0.035	0.03	MOORS AND HEATHLAND -IVR-DSS, grass. Chow (1959) gives a medium value for short grass of 0.030 and 0.035 for high grass.
28	0.085	0.1	0.07	SCLEROPHYLLOUS VEGETATION – average of medium to dense brush in summer (0.07) and winter (0.100) (Chow, 1959). Other literature match IVR-DSS gives 0.045 (Shaw, 1994; www.Imnoeng.com).
29	0.085	0.1	0.07	TRANSITIONAZL WOODLAND-SHRUB - average of medium to dense brush in summer (0.07) and winter (0.100) (Chow, 1959). Other literature match IVR-DSS gives 0.045 (Shaw, 1994; www.Imnoeng.com).
30	0.045	0.0523	0.0377	BEACHES, DUNES, AND SAND PLAINS - river dunes, natural riverbank
31	0.05	0.07	0.03	BARE ROCK - mountains streams, rocky beds (Shaw, 1994)
32	0.0425	0.045	0.04	SPARSELY VEGETATED AREAS - natural water course with some growth and rocks. Chow (1959) and Shaw (1994) give a value of 0.045 and 0.040, respectively.
33	0.033	0.0384	0.0276	BURNT AREAS - natural clean water course
34	0.026	0.0302	0.0218	GLACIERS AND PERPETUAL SNOW - gravel bottom with random stone in mortar (Chow, 1959).
35	0.05	0.0582	0.0419	INLAND MARSHES – (Chow, 1959).
36	0.05	0.0582	0.0419	PEATBOGS – (Chow, 1959).
37	0.045	0.0523	0.0377	SALT-MARSHES - (Chow, 1959).
38	0.045	0.0523	0.0377	SALINES - (Chow, 1959).
39	0.045	0.0523	0.0377	INTERTIDAL FLATS - (Chow, 1959).
40	0.03	0.033	0.027	WATER COURSES - (Chow, 1959; Van der Sande et al., 2003).
41	0.03	0.033	0.027	WATER BODIES - (Chow, 1959; Van der Sande et al., 2003).
42	0.03	0.033	0.027	COASTAL LAGOONS - (Chow, 1959; Van der Sande et al., 2003).
43	0.03	0.033	0.027	ESTUARIES - (Chow, 1959; Van der Sande et al., 2003).
44	0.03	0.033	0.027	SEA AND OCEAN - (Chow, 1959; Van der Sande et al., 2003).
49				'999'
50				'950'
51				'951'
52				'952'

Appendix III Flood Damage Functions

III.1 Damage function from Dutch Ministry of Transport, Public Works and Water Management

Table III.1 *Stage Damage curves of the Dutch Ministry of Transport, Public Works and Water Management (Van der Sande et al., 2003)*

Water depth (m)	Damage functions Rijkswaterstaat (RWS)						
	Agriculture	Purification plant	Roads	Companies	Contents	Structure	House + farm content + structure
0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
0.25	0.25	0.20	0.07	0.03	0.21	0.01	0.07
0.50	0.50	0.30	0.14	0.05	0.35	0.02	0.13
0.75	0.58	0.35	0.21	0.08	0.44	0.04	0.17
1.00	0.64	0.41	0.28	0.10	0.47	0.05	0.18
1.25	0.70	0.47	0.33	0.12	0.48	0.06	0.20
1.50	0.76	0.53	0.37	0.13	0.49	0.08	0.21
1.75	0.82	0.58	0.42	0.15	0.49	0.09	0.22
2.00	0.88	0.64	0.46	0.16	0.50	0.11	0.23
2.25	0.91	0.70	0.51	0.18	0.54	0.16	0.28
2.50	0.93	0.76	0.55	0.19	0.58	0.22	0.33
2.75	0.94	0.80	0.60	0.21	0.62	0.28	0.39
3.00	0.96	0.82	0.64	0.22	0.66	0.35	0.45
3.25	0.98	0.85	0.69	0.32	0.70	0.42	0.51
3.50	1.00	0.87	0.73	0.42	0.74	0.50	0.58
3.75	1.00	0.90	0.78	0.51	0.79	0.59	0.65
4.00	1.00	0.92	0.82	0.61	0.83	0.68	0.73
4.25	1.00	0.95	0.87	0.71	0.90	0.82	0.85
4.50	1.00	0.97	0.91	0.81	0.96	0.92	0.93
4.75	1.00	1.00	0.96	0.90	0.99	0.98	0.98
5.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00
5.25	1.00	1.00	1.00	1.00	1.00	1.00	1.00
5.50	1.00	1.00	1.00	1.00	1.00	1.00	1.00
5.75	1.00	1.00	1.00	1.00	1.00	1.00	1.00
6.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00

The total amount of damage (potential damage, or maximum damage in monetary term) was calculated to the price level of 2004, with the following conversion rate listed in Table III.2.

Table III.2 *Conversion table*

Inflation per year 2.0%	1.02
1EUR = 2.20371 NLG	2.20371
1EUR = 0.6195 GBP	0.6195

Table III.3 *Amount of maximum damage per square meter in 2004*

Land use category	Price level 1993 (Guilder/m²)		Price level 2004 (Euro/m²)	
	Damage	Unit	Damage	Unit
agriculture direct	2.50	m^2	1.41	m^2
agriculture indirect	1.00	m^2	0.56	m^2
greenhouse direct	50.00	m^2	28.21	m^2
greenhouse indirect	20.00	m^2	11.28	m^2
Pavement	3.30	m^2	1.86	m^2
superficial water	0.50	m^2	0.28	m^2
intensive recreation	32.70	m^2	18.45	m^2
Extensive recreation	0.20	m^2	0.11	m^2
highway	50.00	20m width	28.21	m
roads	50.00	10m width	28.21	m
other roads	33.33	6m width	18.81	m
railroads	250.00	3m width	141.05	m
family building/farm	3943.75	$80m^2$ per house	2225.14	piece

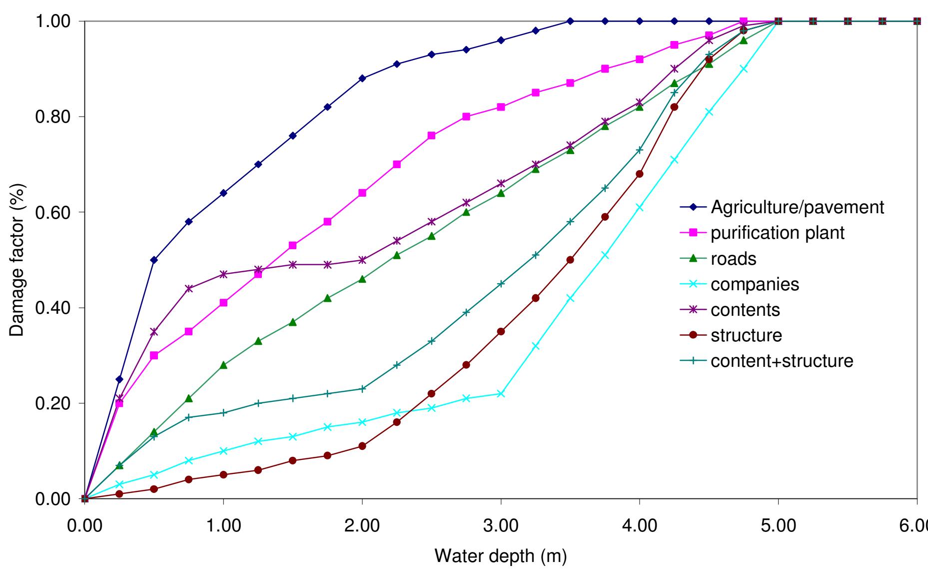

Figure III.1 *Stage damage curves of the Dutch Ministry of Transport, Public Works and Water Management*

III.2 Damage function from HKV consultants

Table III.4 *Stage damage curves of HKV consultants*

Water depth (m)	Greenhouse	Agriculture	Recreation	Total house	Structure	Contents	Industry	Roads
0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
0.25	0.16	1.00	1.00	0.03	0.01	0.12	0.10	0.05
0.50	0.25	1.00	1.00	0.06	0.03	0.24	0.20	0.10
0.75	0.33	1.00	1.00	0.07	0.04	0.35	0.30	0.15
1.00	0.44	1.00	1.00	0.08	0.05	0.47	0.40	0.20
1.25	0.55	1.00	1.00	0.09	0.07	0.48	0.50	0.25
1.50	0.66	1.00	1.00	0.10	0.08	0.49	0.60	0.30
1.75	1.00	1.00	1.00	0.27	0.10	0.49	0.70	0.35
2.00	1.00	1.00	1.00	0.44	0.11	0.50	0.80	0.40
2.25	1.00	1.00	1.00	0.49	0.17	0.54	0.83	0.45
2.50	1.00	1.00	1.00	0.53	0.23	0.58	0.85	0.50
2.75	1.00	1.00	1.00	0.58	0.29	0.62	0.88	0.55
3.00	1.00	1.00	1.00	0.62	0.35	0.66	0.90	0.60
3.25	1.00	1.00	1.00	0.66	0.43	0.70	0.93	0.65
3.50	1.00	1.00	1.00	0.70	0.52	0.75	0.95	0.70
3.75	1.00	1.00	1.00	0.74	0.60	0.79	0.98	0.75
4.00	1.00	1.00	1.00	0.78	0.68	0.83	1.00	0.80
4.25	1.00	1.00	1.00	0.79	0.76	0.87	1.00	0.85
4.50	1.00	1.00	1.00	0.79	0.84	0.92	1.00	0.90
4.75	1.00	1.00	1.00	0.80	0.92	0.96	1.00	0.95
5.00	1.00	1.00	1.00	0.80	1.00	1.00	1.00	1.00
5.25	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00
5.50	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00
5.75	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00
6.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00

Using the same conversion rate as shown in Table III.2, the following table indicates the amount of damage per square meter per land use in 2004 converted from price level 1995 with annual inflation rate of 2%.

Table III.5 *Amount of damage per square meter in 1995 and 2004*

Damage / Land use category	Price level 1995		Transformation to m^2	Price level 1995	Price level 2004
	Unit	Guilder	Unit	(Euro/m^2)	(Euro/m^2)
greenhouse	ha	500000	10000m^2	22.69	27.12
agriculture	ha	2000	10000m^2	0.09	0.11
recreation	ha	500	10000m^2	0.02	0.03
house content	piece	100000	80m^2 per house	567.23	677.89
house structure	piece	200000	80m^2 per house	1134.45	1355.77
industry 'low valued'	ha	1500000	10000m^2	68.07	81.35
rail/road	km	200000	10m width	9.08	10.85

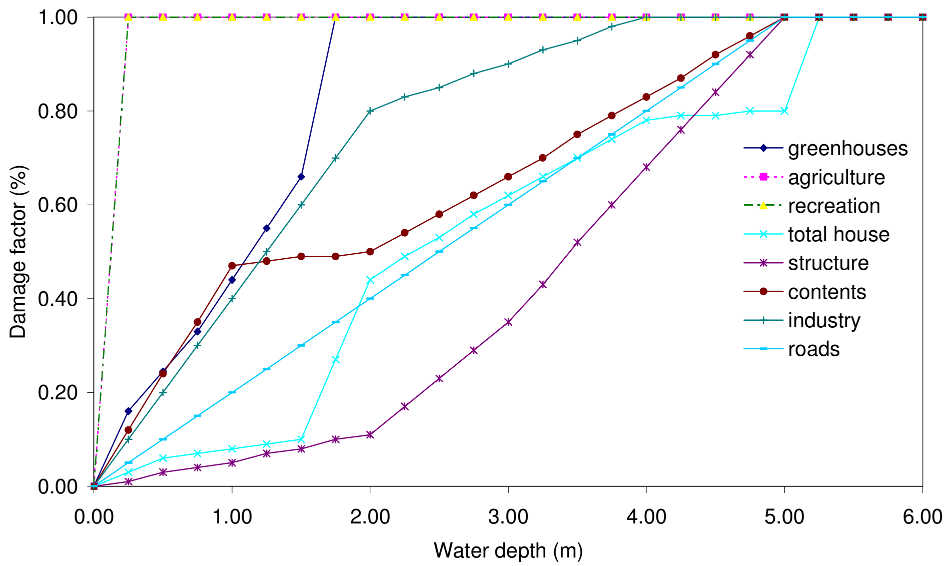

Figure III.2 *Stage damage curves of HKV Consultants*

III.3 Damage function from IKSR (2001)

Table III.6 *Stage ~ damage function, X is the inundation depth (m), Y is damage factor in percentage (%)*

Land use (German)	Land use (English)	Damage function
SF_Siedlung, immobil	Urban area, immobile	$Y=2*X^2 +2*X$
SF_Industrie, immobil	Industry, immobile	$Y=2*X^2 +2*X$
SF_Verkehr, immobile	Traffic, immobile	{0,…,1} Y=10X ab 1 Y=10
SF_Ausruestung Wirtschaft	Infrustructure (business)	Y=11*X + 7.5
SF_Ausruestung wohnen	Infrustructure (residential)	Y= 12*X + 16.25 {X = 1,…, 7}
SF_Ausruestung Staat	Infrustructure (state)	Y=7*X + 5
SF_Siedlung, mobil (35% Wirtschaft, 60% Wohnen, 5% Staat)	Urban area, mobile	Y=11.4*X + 12.625
SF_Industrie, mobil	Industry, mobile	Y= 7*X + 5
SF_Verkehr, mobil	Traffic, mobile	{0,…,1} Y=10X ab 1 Y=10
SF_lwNF	Agriculture	Y = 1
SF_Forst	Forest	Y = 1

Collecting from Rhine Atlas (2001), the maximum damage of each land use in DM/m^2 at price level 2001 (Grossmann, 2004), as shown in table III.7.

Table III.7 *Collecting from Rhine Atlas (2001) (Grossmann, 2004)*

Land use classes		Locations							
Land use in German	Land use in English	Baden-Wuerttemberg		Rheinland-Pfalz		Hessen		Nordrhein-Westfalen	
		Immobile	Mobile	Immobile	Mobile	Immobile	Mobile	Immobile	Mobile
Boden Ackerland (BA)	Agriculture (Farmland)	0.23	0.00	0.23	0.00	0.23	0.00	0.19	0.00
Boden Grunland (BG)	Agriculture (Greenland)	0.10	0.00	0.10	0.00	0.10	0.00	0.07	0.00
Boden Sonderkulturen (BS)	Agriculture	1.73	0.00	1.61	0.00	2.30	0.00	4.27	0.00
Forst (FM)	Forest	3.34	0.00	1.30	0.00	2.58	0.00	1.73	0.00
Energie- und Wasserversorgung, Bergbau (EW)	Urban area (energy, water supply and mining industry)	663.97	6.77	271.85	2.76	473.27	4.83	509.07	3.78
Verarbeitendes Gewerbe, Baugewerbe (VB)	Industry	513.15	169.68	506.06	158.36	504.89	157.12	451.15	156.43
Handel/ Dienstleistungen (HD)	Urban area (service industry)	2602.04	297.47	990.29	93.16	1394.83	114.61	1213.65	138.67
Staat (einschl. Private Organisationen ohne Erwerbscharakter) (ST)	Urban area (country, state)	720.53	7.35	395.99	4.03	484.06	4.94	487.69	2.28
Verkehr und Nachrichtenubermittlung (VN)	Traffic	482.10	4.91	278.72	2.85	586.87	5.99	515.17	3.55
Offentlicher Tiefbau (OET)	Urban area (public mining industry)	135.10	0.00	85.71	0.00	122.01	0.00	183.96	0.00
Wohnkapital (WK)	Urban area	595.49	201.41	531.97	172.14	516.24	183.96	512.29	207.16
Landwirtschaft (einschl. Forstwirtschaft) Gebaude- und Hofflachen (LW)	Agriculture	116.25	10.50	119.00	52.64	92.92	5.86	65.42	0.56
Sonstige (SO)	Others	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00

With the conversion rate of 1EUR = 1.95583 DEM, and inflation per year 2.0%, Table III.7 is converted into Euro/m^2 at price level 2001 as shown in Table III.8.

Table III.8 *Average potential damage of each location for both immobile and mobile component*

Land use (English)	Baden-Wuerttemberg		Rheinland-Pfalz		Hessen		Nordrhein-Westfalen		Average	
	Immobile	Mobile	Immobile	Mobile	Immobile	Mobile	Immobile	Mobile	Immobile	Mobile
Agriculture (Farmland)	0.12	0.00	0.12	0.00	0.12	0.00	0.10	0.00	0.11	0.00
Agriculture (Greenland)	0.05	0.00	0.05	0.00	0.05	0.00	0.04	0.00	0.05	0.00
Agriculture	0.88	0.00	0.82	0.00	1.18	0.00	2.18	0.00	1.27	0.00
Forest	1.71	0.00	0.66	0.00	1.32	0.00	0.88	0.00	1.14	0.00
Urban area (energy, water supply and mining industry)	339.48	3.46	138.99	1.41	241.98	2.47	260.28	1.93	245.18	2.32
Industry	262.37	86.76	258.74	80.97	258.15	80.33	230.67	79.98	252.48	82.01
Urban area (service industry)	1330.40	152.09	506.33	47.63	713.17	58.60	620.53	70.90	792.61	82.31
Urban area (country, state)	368.40	3.76	202.47	2.06	247.50	2.53	249.35	1.17	266.93	2.38
Traffic	246.49	2.51	142.51	1.46	300.06	3.06	263.40	1.82	238.12	2.21
Urban area (public mining industry)	69.08	0.00	43.82	0.00	62.38	0.00	94.06	0.00	67.33	0.00
Urban area	304.47	102.98	271.99	88.01	263.95	94.06	261.93	105.92	275.59	97.74
Agriculture	59.44	5.37	60.84	26.91	47.51	3.00	33.45	0.29	50.31	8.89
Others	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00

Note: Immobile and mobile indicates damage of stationary properties and damage of moveable assets, respectively.

Table III.9 *Summary of IKSR for the maximum damage at price level 2001 in Euro/m^2*

Land use	Maximum Damage price level 2001 Euro/m^2
Urban area	329.53
Industry	252.48
Traffic	238.12
Agriculture	12.93
Forest	1.14
Others	0.00

III.4 Rhine Atlas (2001)

Table III.10 *Stage ~ damage (%) curves used in Rhine Atlas*

Water depth (m)	Urban area (1)	Industry (2)	Traffic (3)	Agriculture (4)	Forest (5)	Others (6)
0.00	0.00	0.00	0.00	0.00	0.00	0.00
0.25	0.03	0.10	0.05	1.00	1.00	0.00
0.50	0.06	0.20	0.10	1.00	1.00	0.00
0.75	0.07	0.30	0.15	1.00	1.00	0.00
1.00	0.08	0.40	0.20	1.00	1.00	0.00
1.25	0.09	0.50	0.25	1.00	1.00	0.00
1.50	0.10	0.60	0.30	1.00	1.00	0.00
1.75	0.27	0.70	0.35	1.00	1.00	0.00
2.00	0.44	0.80	0.40	1.00	1.00	0.00
2.25	0.49	0.83	0.45	1.00	1.00	0.00
2.50	0.53	0.85	0.50	1.00	1.00	0.00
2.75	0.58	0.88	0.55	1.00	1.00	0.00
3.00	0.62	0.90	0.60	1.00	1.00	0.00
3.25	0.66	0.93	0.65	1.00	1.00	0.00
3.50	0.70	0.95	0.70	1.00	1.00	0.00
3.75	0.74	0.98	0.75	1.00	1.00	0.00
4.00	0.78	1.00	0.80	1.00	1.00	0.00
4.25	0.79	1.00	0.85	1.00	1.00	0.00
4.50	0.79	1.00	0.90	1.00	1.00	0.00
4.75	0.80	1.00	0.95	1.00	1.00	0.00
5.00	0.80	1.00	1.00	1.00	1.00	0.00
5.25	0.85	1.00	1.00	1.00	1.00	0.00
5.50	0.90	1.00	1.00	1.00	1.00	0.00
5.75	0.95	1.00	1.00	1.00	1.00	0.00
6.00	1.00	1.00	1.00	1.00	1.00	0.00

III.5 Damage functions used in this case study

After comparing and analysis to different sources of flood damage functions, the stage ~ damage curve provided by Grossmann (2004) is taken

Table III.11 *Stage ~ damage curves (Grossmann, 2004).*

Water depth (m)	Urban area (1)	Industry (2)	Traffic (3)	Agriculture (4)	Forest (5)	Others (6)
0.00	0.00	0.00	0.00	0.00	0.00	0.00
0.50	0.12	0.15	0.05	0.17	1.00	0.00
1.00	0.20	0.20	0.40	0.25	1.00	0.00
1.50	0.24	0.50	0.60	0.35	1.00	0.00
2.00	0.28	0.60	0.60	0.40	1.00	0.00
2.50	0.30	0.70	0.60	0.42	1.00	0.00
3.00	0.36	0.80	0.60	0.44	1.00	0.00
3.50	0.38	0.90	0.60	0.45	1.00	0.00
4.00	0.40	0.90	0.60	0.45	1.00	0.00
4.50	0.40	0.90	0.60	0.45	1.00	0.00

Table III.12 *Potential damage in Euro/m² price level 2001 (Grossmann, 2004)*

Land use code	Land use class	Potential Damage price level 2001 (Euro/m^2)
1	Urban Area	329.53
2	Industry	252.48
3	Traffic	238.12
4	Agriculture	12.93
5	Forest	1.14
6	Others	0.00
7	Waterbody	0.00
8	NODATA	-9999.00
9	Ocean and sea	0.00

Appendix IV Hydraulic Model - HEC6

HEC6 is a 1D steady state hydraulic model ((HEC6 User's Manual, 1993). The main function of HEC6 employed here is to deal with the computation of water surface profiles for steady gradually varied flow in natural or man-made channels. The computational procedure is based on the solution of the one-dimensional energy equation with energy loss due to friction evaluated with Manning's equation, expressed as:

$$WS_2 + \frac{\alpha_2 V_2^2}{2g} = WS_1 + \frac{\alpha_1 V_1^2}{2g} + h_e \qquad \text{(IV.1)}$$

$$h_e = L\bar{S}_f + C_e\left(\frac{\alpha_2 V_2^2}{2g} - \frac{\alpha_1 V_1^2}{2g}\right) \qquad \text{(IV.2)}$$

Where,

WS_1, WS_2	Water level at ends of the modeled river stretch [m]
$i = 1,2$	1- downstream, 2-upstream
V_1, V_2	mean velocities (total discharge/total flow areas) at ends of reach [m/s]
α_1, α_2	Velocity coefficient for flow at ends of reach [-]
g	Gravity acceleration [m/s^2]
h_e	Energy head loss [m]
L	Discharge-weighted reach length [m]
S_f	Representative friction slope for reach [-]
C_e	Expansion or contraction loss coefficient [-]

The unknown water surface elevation at a cross section is determined by an *iterative* solution of Equation (IV.1) and (IV.2) following these steps:

1. Assume a water level at the upstream cross-section WS_2 (or WS_1 at downstream cross section in case of supercritical flow);
2. Determine the corresponding total conveyance ($K = \frac{1.468}{n} ar^{2/3}$), and velocity head;
3. With the value from step 2, compute S_f and solve Equation (2) for h_e;
4. With the value from step 2 and 3, solve Equation (IV.1) for WS_2;
5. Compare the computed value WS_2 with the assumed value in step 1; repeat step1 through 5 until the value agree to within preset tolerance (say 0.01 m).

Appendix V Hydraulic Model - SOBEK1D2D

SOBEK1D2D is a software package that can be used for the simulation of both water quantity and water quality processes in rural and urban areas. It has been developed by WL|Delft Hydraulics (http://wldelft.nl/) with co-operation of DHV (http://www.dhv.nl/). It consists of a model framework in which several available modules can be run in combination as well as stand alone. Among those modules, the flow module and overland flow module are selected for the dike break simulation.

The hydrodynamic modules, either 1D or 2D, or the combination of 1D2D, SOBEK flow modules solves full hydrodynamic models using the so-called "Delft Scheme" (Stelling and Duinmeijer, 2003). It deals with unsteady flow involving all kinds of combination of construction development or changes.

V.1 One-Dimensional (1D) numerical method

In the 1D module, two conservation laws, the shallow water equations are used to describe the mass balance and momentum balance in the flow (e.g. Abbott, 1979; Vreugdenhil, 1994):

$$\frac{\partial h}{\partial t}+\frac{\partial (uh)}{\partial x}=0 \qquad \text{Continuity (V.1a)}$$

$$\frac{\partial u}{\partial t}+u\frac{\partial u}{\partial x}+g\frac{\partial \zeta}{\partial x}+g\frac{u|u|}{C^2 h}=0 \qquad \text{Momentum (V.1b)}$$

Where,

ζ	Water level above plane of reference [m]
u	Velocity [m/s]
t	Time [hr]
x	Distance [m]
g	Gravity acceleration [m/s^2]
h	Total water depth, $h=\zeta+d$,
d	Depth below plane of reference [m]
C	Chézy coefficient [m$^{1/2}$/s]

To solve the above equations numerically, Delft has developed the so-called Delft-scheme (Stelling and Duinmeijer, 2003). The schemes couples two finite difference schematization methods, namely the so-called positive and monotone schemes (Hirsch, 1991) along space (*x* axes), and the so-called θ method, a time weighting coefficient, is applied for the integration along time (*t* axes). The scheme is shown in Figure V.1.

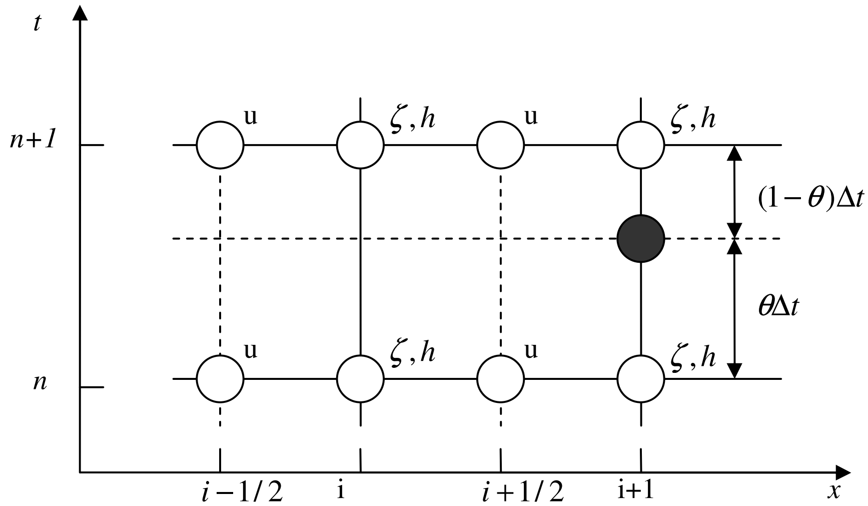

Figure V.1 *Operator of scheme of staggered spatial grid (Hirsch, 1991) along space, and with θ method (Lambert, 1991) along time*

The scheme is applied as:

$$(u_i)_{0x} = \frac{u_{i+1/2} - u_{i-1/2}}{\Delta t}$$

$$(\zeta_{i+1/2}) = \frac{\zeta_{i+1} - \zeta_i}{\Delta x} \qquad \text{(V.2a,b,c)}$$

$$h_{1+1/2}(u) = \begin{cases} h_i, & u > 0 \\ h_{i+1}, & u \le 0 \end{cases}$$

where Δx is the space step, Δt denotes time step.

In turn, Equation V.2a,b can be rewritten in the form:

$$\frac{h^{n+1} - h^n}{\Delta t} + (u^{n+\theta} h^n (u^n))_{0x} = 0 \text{, at } i \qquad \text{(V.3)}$$

$$\frac{u^{n+1} - u^n}{\Delta t} + a(u^n, u^n) + g\zeta_{0x}^{n+\theta} + g\frac{|u^n| u^{n+1}}{C^2 h^n} = 0 \text{, at } i + \frac{1}{2} \qquad \text{(V.4)}$$

where, $u^{n+\theta} = \theta u^{n+1} + (1-\theta)u^n$, and $h_{i+1/2} = (h_i + h_{i+1})/2$. To derive conditions for strict positivity of the water depth, assuming positive flow, Equation 4.8 can be rewritten as:

$$h_i^{n+1} = (1 - \frac{\Delta t \cdot u_{i+1/2}^{n+\theta}}{\Delta x}) h_i^n + \frac{\Delta t \cdot u_{i-1/2}^{n+\theta}}{\Delta x} h_{i-1}^n \qquad \text{(V.5)}$$

with $u_{i+1/2}^{n+\theta} \geq 0, u_{i-1/2}^{n+\theta} \geq 0$

Strict positivity is ensured if:

$$\frac{\Delta t \cdot u_{i+1/2}^{n+\theta}}{\Delta x} < 1 \qquad \text{(V.6)}$$

Simply fulfilling Equation V.5 will prevent wet points from drying, i.e., no special drying and flooding procedures are required for this approach.

The inputs for the 1D module are: a river network with coordinates; river cross-sections in *y-z* profiles (m); bed roughness (Chézy coefficient or Manning number *n*); time series of water level and discharge for representative cross-sections and gauge stations. Through 1D computation, a set of water level and discharge hydrograph are obtained at each calculation node defined during the model set up.

V.2 Two-Dimensional (2D) application

In the overland flow module of SOBEK1D2D, the method described previously is extended to two dimensions. The 2D shallow water equations are given by:

$$\frac{\partial \zeta}{\partial t} + \frac{\partial (uh)}{\partial x} + \frac{\partial (vh)}{\partial y} = 0 \qquad \text{Continuity} \qquad \text{(V.7a)}$$

$$\begin{aligned} \frac{\partial u}{\partial t} + u\frac{\partial u}{\partial x} + v\frac{\partial u}{\partial y} + g\frac{\partial \zeta}{\partial x} + g\frac{u|V|}{C^2 h} = 0 \\ \frac{\partial v}{\partial t} + u\frac{\partial v}{\partial x} + v\frac{\partial v}{\partial y} + g\frac{\partial \zeta}{\partial y} + g\frac{v|V|}{C^2 h} = 0 \end{aligned} \qquad \text{Momentum at } x \text{ and y} \qquad \text{(V.7b)}$$

Where,

u	Velocity in x-direction [m/s]
v	Velocity in y-direction [m/s]
V	Velocity: $V = \sqrt{u^2 + v^2}$ [m/s]
ζ	Water level above plane of reference [m]
C	Chézy coefficient [$m^{1/2}$/s]
d	Depth below plane of reference [m]
h	Total water depth: $\zeta + d$ [m]

The 2D staggered spatial grid is shown in Figure V.2.

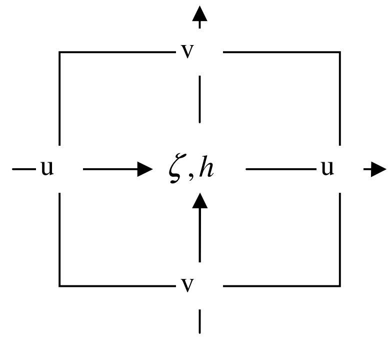

Figure V.2 *Staggered 2D grid*

Applying the 2D staggered spatial grid, the momentum conservative spatial schematization of Equation V.7 reads:

$$\frac{dh}{dt} + (uh(u))_{0x} + (vh(v))_{0y} = 0, \qquad at(i, j) \tag{V.8a}$$

$$\begin{aligned} &\frac{du}{dt} + \frac{1}{\overline{h}^x}((\overline{{}_u q}^x u(\overline{{}_u q}^x))_{0x} + (\overline{{}_v q}^x u(\overline{{}_v q}^x))_{0y} + u\frac{d\overline{h}^x}{dt}) \\ &\quad + g\zeta_{0x} + g\frac{u\|u\|}{C^2\overline{h}^x} = 0, \qquad at(i+1/2, j) \end{aligned} \tag{V.8b}$$

$$\begin{aligned} &\frac{dv}{dt} + \frac{1}{\overline{h}^y}((\overline{{}_u q}^y u(\overline{{}_u q}^y))_{0x} + (\overline{{}_v q}^y u(\overline{{}_v q}^y))_{0y} + v\frac{d\overline{h}^y}{dt}) \\ &\quad + g\zeta_{0y} + g\frac{v\|u\|}{C^2\overline{h}^y} = 0, \qquad at(i, j+1/2) \end{aligned} \tag{V.8c}$$

Where,

$$\begin{aligned} &{}_u q = uh(u) \\ &{}_v q = vh(v) \\ &\overline{h}^x at(i+1/2, j) = \frac{h_{ij} + h_{i+1j}}{2} \\ &\overline{h}^y at(i, j+1/2) = \frac{h_{ij} + h_{ij+1}}{2} \end{aligned} \tag{V.9}$$

all other values are defined accordingly. The momentum conservation characteristics follow easily from multiplying Equation V.8b and Equation V.8c, by $\overline{h}^x$ and $\overline{h}^y$, respectively.

The temporal schematization is implemented with a semi-implicit method following Wilders et al. (1988).

V.3 Dike break simulation using SOBEK1D2D

In SOBEK1D2D, a dike break is simulated in two phases. Starting from a certain moment, the gap crest level is going down with a constant gap width. When a certain maximum depth of the gap is reached, the width of the gap starts increasing. Figure V.3 shows the different stages defined during the growth of the breach:

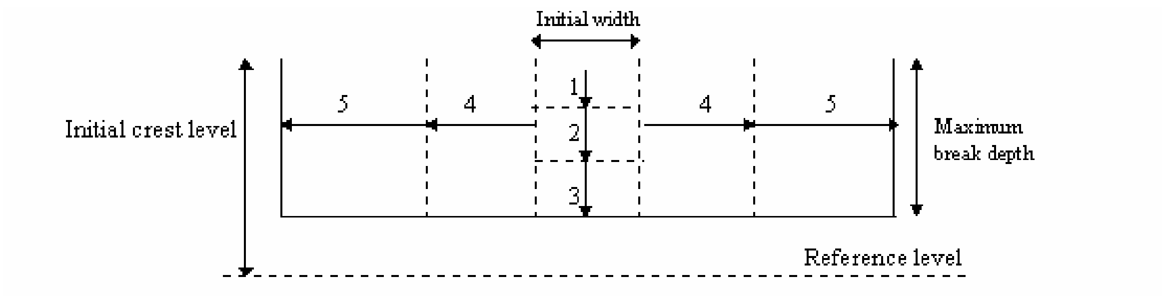

Figure V.3 *Dike break growth mechanism, including vertical development (step 1, 2 and 3) and horizontal development (step 4 and 5)*

The dike break is simulated using the weir formula of Verheij-vdKnaap 2002 (Verheij, 2002), expressed as:

For $t_{start} < t \leq t_0$

$$B(t) = B_0$$
$$Z(t) = z_{crest-level} - (z_{crest-level} - z_{\min}) \times (t / t_0) \qquad \text{(V.10)}$$

For $t > t_0$,

$$B(t) \geq B_0$$
$$B(t_{i+1}) = B(t_i) + \frac{\delta B}{\delta t}\Delta t$$
$$(\frac{\delta B}{\delta t})t_i = \frac{f_1 f_2}{\ln 10}\frac{(g(h_{up} - h_{down}))^{1.5}}{u_c^2}\frac{1}{1 + \frac{f_2 g}{u_c}(t_1 - t_0)} \qquad \text{(V.11)}$$

Where,

B_0	Initial width of the breach [m]
$B(t)$	Width of the breach at point in time t [m]
t_{start}	Point-in-time at which the breach starts to develop
$t_0 = t_{start} + T_0$	The point-in-time when the maximum breach-depth (z_{min}) is reached
T_0	Time span over which the breach having a constant initial width (B_0) is lowered from its initial crest level (z_{crest}-*level*) to its final crest level (z_{min}) [hr]
f_1	Factor1, constant factor (input parameter) [-]
f_2	Factor 2, constant factor (input parameter) [-]
g	Gravitational acceleration [m/s^2]

h_{up}	Upstream water level at point-in-time t [m]
h_{down}	Downstream water level at point-in-time t [m]
u_c	Constant critical flow velocity sediment/soil (input parameters) [m/s]
$z(t)$	Elevation of the dike-breach at point-in-time t [m]
$z_{crest\text{-}level}$	Elevation of the crest-level of the dike at $t = t_{start}$ (input parameter) [m]
z_{min}	Elevation of the dike-breach at $t = t_0$ (input parameter) [m]

Empirical parameters are used in the formula Verheij-vdKnaap 2002, as listed in Table V.1

Table V.1 *Default parameters used in the Verheij-vdKnaap2002 formula (Verheij, 2002)*

Parameter	Default	Range
f_1	1.3	0.5 - 5
f_2	0.04	0.01 - 1
B_0	10	1 - 100 m
T_0	0.1 hours	0.1 - 12 hours
u_c	0.2 m/s	0.1 - 10 m/s

Using Equation V.10 and V.11, h_{down} will never be lower than z_{min}. The value of u_c is defined associated with the material or soil type the dike was constructed with. For the dike near Sandau, compacted clay is assumed.

Appendix VI Flood Routing Coefficients – ELBA

Location			Discharge regime								
Upstream	Downstream	River length (km)	High			Medium			Low		
			Q1	D1	v1	Q2	D2	v2	Q3	D3	v3
Usti	Schona	45.3	50000	0	0	7000	37.38	8.951	1000	21.7	6.885
Usti	Dresden	98.8	50000	132.87	8.436	1000	140.29	8.085	300	78.48	7.197
Dresden	Riesa	52.8	50000	70.62	5.94	1600	70.62	5.94	800	67.3	5.56
Dresden	Torgau	99	50000	82.4	3.27	1600	70.62	5.94	800	67.3	5.56
Torgau	Twitteberg	59.5	50000	10.3	2.09	1400	14.28	2.19	700	32.11	2.799
Wittenberg	Dessau(Ross.)	45.4	50000	0	0	7000	3.5	1.744	1000	11.32	3.384
Witenberg	Dessau	47.1	50000	0	0	7000	3.5	1.744	1000	11.32	3.384
Golzern	Bad Duben	57	50000	0	0	7000	59.8	2.96	250	28.56	4.53
Bad Duben	Muldemundg	64	50000	18.21	1.331	750	11.85	2.051	250	57.2	2.889
Dessau	Aken	15.2	50000	0	0	7000	3.5	1.744	1000	11.32	3.384
Zeitz	Leipzig	46				7000	13.94	2.035	130	15.75	2.794
zeitz (+Bohlen)	Halle	110				7000	13.94	2.035	130	15.75	2.794
Naumburg	Halle	66				7000	7.33	2.388	200	33.93	3.828
Halle	Bernburg	58				7000	27	2.5	300	17.9	6.5
Halle	Calbe	78				7000	39.2	2.56	300	20.5	6.8
Hadmersleben	Calbe	56				7000	15	1.7	60	7.45	2.54
Calbe	Barby	19.3				7000	39.2	2.56	300	20.5	6.8
Aken	Barby	18.7	50000	0	0	7000	8.86	2.21	1340	8.91	3.338
Barby	Magdeburg	31.2	50000	33.1	3.22	2400	33.4	2.22	1340	27.9	3.15
Barby	Niegripp	50.2	50000	33.1	3.22	2400	33.4	2.22	1340	27.9	3.15
Barby	Tangermunde	94.8	50000	33.1	3.22	2400	33.4	2.22	1340	27.9	3.15
Tangermunde	Wittenberge	66.7	50000	33.2	2.28	2400	33.2	2.28	1290	132.8	2.67
Wittenberge	Lenzen	29.8	50000	0	0	7000	33.2	2.28	1640	132.8	2.67
Wittenberge	Domitz	49.5	50000	0	0	7000	33.2	2.28	1640	132.8	2.67
Wittenberge	Neu Darchau	80.9	50000	0	0	7000	35.6	2.4	1640	64.2	2.56
Neu darchau	Boizenburg	23.7	50000	0	0	7000	35.6	2.4	1640	64.2	2.56

Note: Three discharge regimes are distinguished as *high*, *medium* and *low* (Fröhlich, 1998; Busch et al., 1999).

Summary

There is evidence of a growing gap between the available scientific knowledge and the practical needs of Integrated River Basin Management (IRBM). This gap becomes manifest when e.g. water management objectives or problems encountered by water users are not clearly defined and correctly translated into the different "languages" that are used by scientists or end users. Or when the choice of measures is not based on adequate scientific principles but on socio-political preferences or convenience, which can result in taking ineffective measures. Moreover, the existence of sometimes conflicting objectives present at different management levels increases the complexity of IRBM. This calls for reliable and efficient tools to support the decision making process: Decision Support Systems (DSSs). Preferably, such instruments should not be excessively complex nor overly simple, but appropriate for analyzing the problems and for assessing different promising management alternatives, by stimulating the interaction between the various stakeholders, decision makers and modelers.

However, the design of a DSS remains difficult due to (i) the lack of a methodology for selecting/formulating appropriate models that are neither excessively complex nor overly coarse with respect to their functionality requirement, (ii) the lack of an evaluation approach to measure the overall performance of a DSS including the effects of uncertainty associated with the decision variables. Solving these two problems is essential for the design of a DSS. The difficulties encountered in the design of a DSS for IRBM can be found in any design that includes multiple objectives involving various disciplines and covering different temporal and spatial scales.

From the design point of view, most DSSs for IRBM are based on a systems analysis approach. This approach begins with the study of the physical conditions, and then defines clearly the management objectives; finally the system is formulated in terms of functionality requirements. Previous DSS design approaches can be categorized into *user-oriented* versus *knowledge-driven* approaches, each of which leads to a different design architecture. The user-oriented method aims at a DSS serving a specific problem for a specific river basin, whereas the knowledge-driven method aims to develop a DSS as a generic tool to be used in an arbitrary river basin to deal with a range of problems. The model selection methods are influenced by the design approach chosen. The user-oriented approach tends to prefer tailored models and data, and sometimes suffers from a lack of sufficient scientific principles, because of the ad-hoc design choices. On the other hand, the knowledge-driven approach aims to incorporate generic mathematical models that can be readily adjusted when possible changes in the requirements occur, which can result in an overly complex design. Nevertheless, the question "*How to select models with a minimum but adequate degree of complexity?*" has not been well answered for both methods.

A comparative approach, i.e. selecting a model based on its performance according to certain specified criteria, is commonly used when a model can be selected from a choice of candidate models. However, such a situation is rare in DSS design. In practice models are often selected based on qualitative analysis by studying the phenomena that require a certain type of mathematical model – or simply by the availability of a model. Comparing model performance is a scientific research activity rather than a common step in practice. Secondly, the performance of

a model is user-dependent and very much related to experience. Moreover, the choice of what models can be considered appropriate occurs at different levels of analysis. For example it can pertain to a single objective such as flood safety or, when multiple objectives are considered, to the tradeoff between safety and nature quality. Hence, in practice it is difficult to conclude that one model is more appropriate than another.

Therefore, to obtain an appropriate DSS, two research questions need to be answered: (i) how to determine appropriate model/system complexity? and (ii) how to obtain and evaluate a DSS's overall performance? These questions are addressed by the framework for DSS design, which is proposed in this thesis following a systems analysis approach. The framework consists of an iterative and interactive process comprising two phases: *qualitative analysis* and *quantitative analysis*. Qualitative analysis involves: problem definition, study of physical conditions, followed by identifying the causal relationships connecting relevant processes and variables to the objectives and measures; this results in a conceptual design using the double-direction search method. Quantitative analysis focuses on the formulation of models, model calibration and validation, as well as uncertainty analysis.

In the proposed framework for the design of a DSS, the problem of how to select an appropriate model is addressed by a *double-direction search* method (Chapter 2). This method determines the required model complexity and suitable modeling systems available through causal reasoning from two ends, viz. guided by the predetermined environmental and socio-economic indicators resp. The appropriate level of complexity is determined by two search directions, viz. the forward-search direction and the backward-search. Forward-search involves the physical problem identification which determines what disciplinary processes are involved (e.g. hydraulics, hydrology or ecology); the backward-search direction prevents divergence in complexity by using criteria such as the allowable water level or acceptable flood risk.

Another issue of how to obtain and evaluate a DSS's overall performance is addressed by the use of Uncertainty Analysis (UA). Amongst the various approaches, UA is commonly found to be an important methodology for evaluating DSS performance. UA can provide insight into uncertainty contributions from different system components. This can provide comprehensive presentations of the system outcome to aid decision making. However, there are a number of difficulties related to UA for DSS, particularly the propagation of model uncertainty through an integrated system of multiple component models – this has not been observed in the literature so far. This issue has been addressed by the use of the scenario tree method adopted in this thesis.

Using the proposed steps involved in the design of a DSS, based on the management objectives of a DSS for River Elbe in Germany, a framework for Integrated Flood Risk Assessment (IFRA) is obtained. A case study was carried out using the IFRA framework to the region near the town of Sandau, along the German river Elbe, assessing both local and non-local effects.

In the history of FRA, two approaches are often encountered in Flood Risk Assessment (FRA) that can be used for different purposes, namely the *statistical risk-analysis based approach* and the *physically-based approach*. Based on water level ~ discharge relationships, the statistical approach assesses flood risk using expected annual damage, which can be used to support long-term planning such as the construction of a dike. It is usually applied at both large (size of several hundred kilometers) and small spatial scale (e.g. <100 km). The physically-based approach calculates the flood damage for specific flood events using, preferably, 2D-hydrodynamic inundation models. This method gives insight into the damage associated with a certain flood event and the change of physical conditions, information that can be used to support short-term management decisions such as the operation of a retention basin during flooding. Currently, it has been found that due to the large data demand and computation involvement, the physically-based FRA is more applied at the smaller spatial scales (e.g. <100 km).

To achieve effective flood management, all short-term operational activities should be embedded within the long-term planning strategy of the river basin. Inversely, at the regional scale (normally around 50 km) possible overall effects should be taken into account. Thus, decision support tools for IFRA are needed that can handle different temporal and spatial scales. Such an IFRA can be developed based on the two flood risk assessment approaches described above (viz. statistical and physically-based, resp.).

Current FRA methods are found to be limited when neglecting flow velocity effects, which can be clearly significant, given examples such as the 2004 tsunami in South Asia. The effect of flow velocity on flood risk has not been quantitatively expressed or systematically accounted for in classical risk assessment methods so far. To include the effect of flow velocity, a so-called risk matrix (Chapter 3) has been used. The risk level concept used in the risk matrix combines inundation damage and damage due to velocity, corresponding to the direct actions taken in response to flooding. The risk levels are defined in consultation with end-users according to, for example, socio-economic criteria, and need not be applicable to other regions.

Another issue in IFRA is that in general time is an item of concern in decision making, for example 1-2 hours warning time is the minimum in case of an emergency. However, most inundation modeling is based on 2D hydrodynamic models, which may require longer computing times. This problem can be overcome by invoking parallel computing or by using Artificial Neural Network emulation of pre-computed results, but always depends on the availability of time needed for model set up, calibration and validation. These difficulties limit the flexibility of a 2D model when end users require a rapid FRA. Moreover, data demands are usually large and difficult to satisfy. Hence the challenge is to obtain a rapid FRA without the need for carrying out large computations. Clearly, any such rapid approach can only be an approximation.

To investigate its applicability, the conceptual framework of IFRA has been applied using case studies at the *local* level (Chapter 4) and at the *non-local* level (Chapter 5), within the region around Sandau. At the local scale the risk/damage is assessed for the flooded area without considering the impact on downstream areas. At the non-local scale FRA is aimed at risk mitigation, by assessing flood management activities like deliberate dike breaching at an upstream area of lower economic importance.

As an example of local FRA, two scenarios have been studied for each risk assessment approach: a scenario with and without the presence of a dike following the *statistical* approach, and a scenario with and without a dike break using the *physically-based* approach. The results show a clear difference between the scenarios. The effect of uncertainty is demonstrated in the implementation of flood management measures both for the scenarios based on the statistical approach (in the presence and absence of a dike), and for the scenarios based on the physically-based risk model (with an intentional dike break aiming at risk mitigation). The uncertainty distribution of each risk indicator shows that the effects of these measures are distinguishable, which means the DSS can be considered appropriate to support decision making.

For non-local FRA, a case study has been carried out using the physically-based hydrodynamic model SOBEK1D2D to simulate the inundations. Scenarios have been explored to estimate the impact of an artificial dike break. The results point to an effective mitigation of the downstream flood risk by an intentional dike break upstream, provided the moment and width of the breach are carefully chosen.

To evaluate the performance of the physically-based FRA, UA has been carried out to study the effect of uncertainty related to the most sensitive parameters, viz. elevation data and roughness coefficients. For the elevation data, a comparison has been made of the effect of different aggregation methods used to lower the spatial resolution of the elevation mesh, e.g. to reduce computational limitations. To determine the contribution of the uncertainty in the hydraulic roughness, a range of results has been obtained by varying the roughness associated with each land use. The results show a significant contribution to the overall uncertainty in flood damage caused by the aggregation of the elevation data, and a very small contribution from roughness, when the depth ~ damage functions are applied. However, a significant effect of roughness was observed in the uncertainty in the flow velocity. This indicates the significance of the contribution of the flow velocity to the flood damage. The results also indicate the usefulness of UA to understand the effect of imperfect data and approximate models to the overall uncertainty in the outcome of the DSS.

There are also technical issues that need to be dealt with in UA, in particular when large computational loads are required, for example, when two or three dimensional hydrodynamic modeling is needed. To reduce computing time, a simplified Monte Carlo simulation - the *scenario tree method* (Chapter 2) - is used in this thesis. This method can be used to effectively propagate uncertainties from the hydraulic models to the risk indicators without the need to carry out a large number of model runs, by assuming self-similarity in the uncertainty distributions.

The selection of hydraulic models for IFRA has been found to depend on the management objective. For rapid assessment, an approximation of the fully 2D approach can be obtained using GIS technology in combination with 1D flood routing in the river channel. The GIS-based approach, proposed in this thesis (Chapter 5), is such a method. The results agree well with those obtained with SOBEK1D2D in terms of the maximum inundation depth, flooding area, and economic loss due to inundation. However, approximating the damaging effect of flow velocity using GIS technology alone – i.e. without taking the full dynamics into consideration – turns out to be unsatisfactory. This may become a new challenge for future research related to IFRA.

Samenvatting

Er blijkt een groeiende kloof te bestaan tussen de beschikbare wetenschappelijke kennis en de praktische behoeften voor Integraal Rivier Beheer (IRB). Deze kloof manifesteert zich bijvoorbeeld door het feit dat de doelstellingen voor waterbeheer of de problemen, die door de gebruikers worden ondervonden, niet duidelijk zijn gedefinieerd en correct vertaald in de verschillende "talen" die door de wetenschappers en eindgebruikers worden gehanteerd. Of, wanneer de keuze van maatregelen niet gestoeld is op toereikende wetenschappelijke principes, maar op sociaal-politieke voorkeuren of praktische overwegingen. Dit kan leiden tot een keuze voor ineffectieve maatregelen. Bovendien neemt de complexiteit van IRB toe door de aanwezigheid van soms conflicterende doelstellingen op verschillende beheersniveaus. Dit alles vraagt om betrouwbare en efficiënte instrumenten om het besluitvormingsproces te ondersteunen, ofwel Beslissings Ondersteunende Systemen (BOS). Bij voorkeur dienen deze instrumenten niet overmatig gecompliceerd of vereenvoudigd te zijn, maar juist geschikt voor de analyse van de problemen en de mogelijk geschikte beheersalternatieven, door de interactie tussen de diverse belanghebbenden, besluitnemers, en modelleurs te stimuleren.

Echter, het ontwerp van een BOS is nog steeds lastig vanwege (i) het ontbreken van een methodologie voor de selectie en het formuleren van geschikte modellen, die noch overmatig complex zijn, noch overmatig vereenvoudigd met betrekking tot de functionele eisen, en (ii) het ontbreken van een aanpak om de prestatie van het BOS als geheel te kunnen meten, waarbij ook de gevolgen van onzekerheid in de beslissingsvariabelen in acht genomen worden. Voor het ontwerp van een DSS is het essentieel dat deze twee problemen zijn opgelost. De moeilijkheden die zich bij het ontwerp van een BOS voor IRB voordoen komt men tegen bij elk ontwerp, waarbij sprake is van meerdere doelstellingen die betrekking hebben op verschillende vakdisciplines met verschillende tijd- en ruimteschalen.

Ontwerptechnisch zijn de meeste BOSen voor IRB gebaseerd op een systeemanalytische benadering. Deze vangt aan met een studie van de fysische omstandigheden, waarna duidelijke beheersdoelstellingen worden geformuleerd, en tot slot het systeem wordt opgezet volgens de functionele eisen. Eerdere BOS-ontwerpen kunnen worden ingedeeld in *gebruikersgeoriënteerde* en *kennisgeoriënteerde* benaderingen, die elk tot een andere systeemarchitectuur leiden. De gebruikersgeoriënteerde methode is gericht op het ontwerp van een BOS voor een specifiek probleem of een bepaald stroomgebied, terwijl de kennisgeoriënteerde methode zich richt op de ontwikkeling van een BOS als generiek instrument voor willekeurige stroomgebieden en een scala aan problemen. De modelselectiemethoden zijn afhankelijk van de gekozen benadering voor het ontwerp. Bij de gebruikersgeoriënteerde methode is men geneigd om de voorkeur te geven aan maatwerkmodellering en -gegevens, wat soms leidt tot een gebrek aan wetenschappelijke onderbouwing, omdat keuzes ad-hoc worden gemaakt. Anderzijds richt de kennisgeoriënteerde methode zich er op generieke mathematische modellen, die eenvoudig zijn aan te passen in geval van veranderende functionele eisen, in het ontwerp op te nemen, wat tot een overmatig complex ontwerp kan leiden. Derhalve is de vraag "*Hoe kunnen modellen met een minimale maar toereikende graad van complexiteit worden gekozen?*" voor beide benaderingen nog niet beantwoord.

Een vergelijkende benadering, d.w.z. dat een model wordt gekozen op basis van de prestaties volgens bepaalde criteria, wordt gewoonlijk gebruikt indien er een keuzemogelijkheid is uit meerdere kandidaat-modellen. Een dergelijke situatie is echter zeldzaam bij het ontwerp van een BOS. In de praktijk worden modellen vaak gekozen op grond van kwalitatieve analyse - door de verschijnselen die een bepaald type model vereisen te bestuderen - of eenvoudigweg op basis van beschikbaarheid van het model. Het vergelijken van modellen aan de hand van prestatiecriteria is eerder een wetenschappelijke dan een praktische aangelegenheid. Bovendien is de prestatie van een model gebruikersafhankelijk en wordt deze sterk bepaald door de ervaring van de modelleur. De keuze voor geschikte modellen vindt plaats op verschillende niveaus van analyse, en kan bijvoorbeeld betrekking hebben op een enkelvoudige doelstelling zoals de overstromingsveiligheid of, indien meerdere doelstellingen in beschouwing worden genomen, de afweging tussen veiligheid en natuurkwaliteit. Daarom is het in praktische zin nog steeds moeilijk om aan te geven dat een model meer geschikt is dan een ander model.

Om tot een geschikt BOS te komen dienen twee onderzoeksvragen te worden beantwoord: (i) hoe kan het geschikte complexiteitsniveau van een model/systeem worden bepaald? en (ii) hoe kan men de kwaliteit van een BOS als geheel vaststellen? Het raamwerk voor het ontwerpen van een BOS, dat in dit proefschrift wordt voorgesteld om deze vragen te beantwoorden, is gebaseerd op een systeemanalytische aanpak. Het raamwerk is gebaseerd op een iteratieve en interactieve ontwerpprocedure die uit twee fasen bestaat: *kwalitatieve analyse* en *kwantitatieve analyse*. De kwalitatieve analyse omvat achtereenvolgens de probleemdefinitie, het bestuderen van de fysische omstandigheden, het identificeren van de oorzaak-gevolgrelaties, die de relevante processen en variabelen verbinden met de doelstellingen en maatregelen, en wordt afgesloten met een conceptueel ontwerp op basis van een *twee-richtingszoekmethode*. De kwantitatieve analyse richt zich op het opstellen van modellen, de modelkalibratie en modelvalidatie, alsmede de onzekerheidsanalyse.

Het centrale probleem bij de kwalitatieve fase is de keuze van geschikte modellen. In dit proefschrift wordt dit probleem benaderd door een *zoekmethode in twee richtingen* (Hoofdstuk 2). Met deze methode kunnen het vereiste complexiteitsniveau van een model en de beschikbaarheid van geschikte modelleerplatformen worden vastgesteld aan de hand van oorzaak-gevolgredeneringen uit twee richtingen, uitgaande van de vooraf bepaalde milieu- en sociaal-economische indicatoren. De geschikte complexiteit wordt vanuit twee zoekrichtingen, namelijk voorwaarts en achterwaarts, vastgesteld. Bij de voorwaartse zoekmethode gaat uit van de fysische probleemdefinitie, die bepalend is voor de relevantie van vakdisciplines zoals bijv. hydraulica, hydrologie, of ecologie. De achterwaartse zoekmethode is gericht op het voorkomen van divergerende modelcomplexiteiten door de toepassing van criteria zoals het toegestane hoogwaterniveau of het acceptabele overstromingsrisico.

Nadat een BOS met de gekozen modellen is ontwikkeld, is de volgende stap om het ontwerp op basis van de prestaties van het BOS in zijn totaliteit te toetsen. Uit de keuze van mogelijke benaderingen voor de evaluatie van een BOS wordt Onzekerheids Analyse (OA) gewoonlijk beschouwd als een belangrijke methode. Er doen zich bij de OA van een BOS echter een aantal problemen voor die vooral betrekking hebben op de voortplanting van modelonzekerheden door het geïntegreerde systeem van modelcomponenten, en tot nu toe niet in de wetenschappelijke literatuur zijn opgemerkt. Hiertoe wordt de scenarioboommethode toegepast. In dit proefschrift wordt een raamwerk voor het ontwerp van een BOS voor de Integrale Overstromings Risico Analyse (IORA) voorgesteld, en toegepast op het gebied bij de stad Sandau, langs de rivier de Elbe in Duitsland, zowel met betrekking tot de lokale als de niet-lokale effecten van maatregelen.

Men komt bij de Overstromings Risico Analyse (ORA) twee benaderingen tegen, die voor verschillende doelstellingen kunnen worden ingezet, namelijk de *statistische risicoanalyse* en de *fysische risicoanalyse*. Bij de statistische aanpak wordt het overstromingsrisico bepaald om de analyse van langetermijnoplossingen, zoals de aanleg van een dijk, te ondersteunen. Gewoonlijk wordt deze benadering toegepast op grotere schaalniveaus (ter grootte van honderden kilometers). Bij de fysische aanpak wordt het overstromingsrisico voor specifieke hoogwatergebeurtenissen berekend, bij voorkeur d.m.v. 2D-hydrodynamische modellering van de inundatie. Met deze methode kan inzicht verkregen worden in de schade die samenhangt met een bepaalde hoogwatergebeurtenis en verandering in de fysieke omstandigheden, informatie die gebruikt kan worden ter ondersteuning van kortetermijnbeheersbeslissingen, zoals de inzet van een retentiegebied tijdens een overstroming. Vanwege de grote behoefte aan gegevens en omvang van de berekeningen wordt de fysische benadering vooral op kleinere schaalniveaus (< 100 km) toegepast.

Voor de effectiviteit van overstromingsbeheer is het noodzakelijk dat alle operationele, kortetermijnmaatregelen ingebed zijn in de langetermijnstrategie voor het beheer van het stroomgebied. Omgekeerd dienen op de regionale schaal (normaal gesproken rond de 50 km) de mogelijke gevolgen in hun totaliteit in beschouwing genomen te worden. Dit betekent dat beslissingsondersteunende systemen, die om kunnen gaan met verschillende tijd- en ruimteschalen, nodig zijn voor IORA. Een dergelijke IORA kan worden ontwikkeld op basis van de twee genoemde risicoanalysemethoden (respectievelijk de statistische en fysische benadering).

De bestaande ORA methoden hebben hun beperking omdat de, duidelijk belangrijke, effecten van stroomsnelheden niet in beschouwing genomen worden, zie ook het voorbeeld van de tsunami van 2004 in Zuidoost Azië. Het effect van de stroomsnelheid op het overstromingsrisico is nog niet gekwantificeerd of systematisch in beschouwing genomen in de bestaande risico analyse methoden. Om het effect van de stroomsnelheid bij de analyse mee te nemen wordt in dit proefschrift een zogenaamde risicomatrix gebruikt (Hoofdstuk 3). In deze matrix wordt een risiconiveau vastgelegd aan de hand van de combinatie van de overstromingsschade en het effect van de stroomsnelheid op de schade, gekoppeld aan de directe reactie op een overstroming. De risiconiveaus worden in overleg met de eindgebruikers vastgesteld op basis van, bijvoorbeeld, sociaal-economische criteria, en hoeven niet noodzakelijkerwijs toepasbaar te zijn voor andere gebieden.

Een andere kwestie die bij IORA speelt, is dat in het algemeen de tijd een factor van belang is voor de besluitvorming. Zo is een minimale waarschuwingstijd van 1-2 uur bijvoorbeeld noodzakelijk in geval van een noodsituatie. De meeste inundatiemodellen zijn echter gebaseerd op 2D hydrodynamische modellen, die wezenlijke langere rekentijden vereisen. Dit probleem kan worden opgelost door parallelle berekeningen of door met Kunstmatige Neurale Netwerken voorberekende resultaten te gebruiken, maar altijd zal de beschikbaarheid van tijd voor het opzetten van het model, modelkalibratie en modelvalidatie van invloed zijn. Deze moeilijkheden verminderen de flexibiliteit van 2D modellen wanneer de eindgebruikers een snelle ORA verlangen. Bovendien is de behoefte aan gegevens van dergelijke modellen gewoonlijk groot, en moeilijk om aan te voldoen. Daarom is het een uitdaging om een snelle methode voor ORA, zonder de noodzaak van grootschalige berekeningen, te ontwikkelen. Het zal duidelijk zijn dat een dergelijke snelle aanpak slechts een benadering van de werkelijkheid kan zijn.

Uitgaand van het raamwerk voor het ontwerpen van een BOS, zoals voorgesteld in Hoofdstuk 2, is een conceptueel raamwerk voor IORA op basis van kwalitatieve analyse opgezet (Hoofdstuk 3). Om de toepasbaarheid van dit raamwerk te onderzoeken zijn voor het gebied rond Sandau kwantitatieve analyses uitgevoerd aan de hand van case studies op het *lokale* (Hoofdstuk 4) en het *niet-lokale* schaalniveau (Hoofdstuk 5). Op het lokale schaalniveau worden risico en/of schade vastgesteld voor het overstromingsgebied, zonder dat de benedenstroomse gevolgen in acht genomen worden. Op het niet-lokale schaalniveau is de ORA er op gericht het risico te verminderen door hoogwatermaatregelen, zoals het opzettelijk door laten breken van een dijk op een bovenstroomse locatie die van minder economisch belang is, te analyseren.

Als voorbeeld van een lokale ORA zijn twee scenario's bestudeerd voor elk van de twee risico-benaderingsmethoden: voor de *statistische* aanpak een scenario met en zonder dijk, en voor de *fysische* benadering een scenario met en zonder dijkdoorbraak. De resultaten voor de scenario's vertonen duidelijke verschillen. Het effect van onzekerheid op de implementatie van de overstromingsmaatregelen wordt zowel voor de scenario's met de statistische aanpak (wel of geen dijk) als voor de scenario's met de fysische aanpak (wel of geen opzettelijke dijkdoorbraak om het overstromingsrisico te verminderen) aangetoond. De onzekerheidsverdeling van elke risico-indicator laat zien dat de gevolgen van deze maatregelen onderscheidbaar zijn, wat betekent dat het BOS geschikt is om de besluitvorming te ondersteunen.

Met het fysisch-georiënteerde hydrodynamische model SOBEK1D2D is een casestudie van een niet-lokale ORA op basis van simulatie van de inundaties uitgevoerd. Aan de hand van scenario's is onderzocht wat het effect van een kunstmatige dijkdoorbraak zou zijn. De resultaten wijzen op een effectieve vermindering van het benedenstroomse overstromingsrisico door een bovenstroomse, opzettelijke, dijkdoorbraak, vooropgesteld dat het moment en de grootte van de dijkbraak met zorg zijn gekozen.

Om de prestatiewaarde van de fysisch georiënteerde ORA te kunnen vaststellen is een OA uitgevoerd, waarbij de bijdrage aan de onzekerheid van de meest gevoelige parameters, de landhoogte en ruwheidcoëfficiënten, is bepaald. Voor de landhoogte is een vergelijking gemaakt van de invloed van verschillende aggregatiemethoden ter verkleining van het oplossend vermogen van het hoogteraster, met het oog op beperking van de rekentijden. Om het gevolg van de onzekerheid in de hydraulische ruwheid te bepalen zijn de resultaten, die worden verkregen met verschillende ruwheidswaarden gekoppeld aan het landgebruik, vergeleken. Uit de resultaten blijkt dat, indien van de inundatiediepte afhankelijke schadefuncties worden toegepast, de bijdrage van de aggregatiemethode voor de landhoogte aan de onzekerheid in de overstromingsschade significant is, terwijl de bijdrage van de ruwheid klein is. De bijdrage van de ruwheid aan de onzekerheid in de stroomsnelheid blijkt echter groot te zijn. Dit wijst er op dat de bijdrage van de stroomsnelheid aan de overstromingsschade van belang is. De resultaten laten ook zien dat OA bruikbaar is om de gevolgen van onvolkomenheden in de gegevens en modelbenaderingen voor de onzekerheden in de uitkomsten van een BOS te begrijpen.

Met OA hangen ook een aantal technische kwesties samen, die te dienen worden opgelost, vooral indien grootschalige berekeningen, bijvoorbeeld voor twee- of driedimensionale hydrodynamische modellering, nodig zijn. Om de rekentijden te verminderen is in dit proefschrift een vereenvoudigde Monte Carlo simulatie toegepast, de *scenarioboommethode* (Hoofdstuk 2). Deze methode kan, door gelijksoortigheid van de onzekerheidsverdelingen aan te nemen, effectief worden ingezet om onzekerheden van de hydraulische modellen naar de risicoindicatoren voort te planten, zonder dat een groot aantal modelsimulaties hoeft te worden uitgevoerd.

De keuze van hydraulische modellen voor IORA blijkt bepaald te worden door de beheersdoelstelling. Voor de snelle analyse van maatregelen (rapid assessment), kan een benadering van de volledige 2D aanpak worden bereikt door GIS-technologie te combineren met 1D voortplanting van een hoogwatergolf in de hoofdgeul. De GIS-gebaseerde aanpak, die in dit proefschrift wordt voorgesteld (Hoofdstuk 5), is een voorbeeld hiervan. In termen van de maximale inundatiedieptes, het overstroomde oppervlak, en de economische gevolgen van de inundatie, stemmen de resultaten goed overeen met die van SOBEK1D2D. De benadering van de schadelijke gevolgen van stroomsnelheden op basis van uitsluitend GIS technieken - d.w.z. zonder dat de volledige dynamica in beschouwing wordt genomen - blijkt echter onbevredigend te zijn. Dit kan een nieuwe uitdaging zijn voor toekomstig onderzoek in relatie tot IORA.

内容摘要 (Summary in Chinese)

在综合河流管理(IRBM)中普遍存在着科学与实践之间的距离。当IRBM的管理目标或问题没有被准确翻译为科学语言，或者管理措施的选择没有基于科学原理而是为了达到某种社会或政治目的时，这个距离尤其明显。此外，多种相互矛盾的管理目标的同时考虑也增加了IRBM的复杂性。这些问题的存在需要一个可靠有效的决策支持系统(DSS)来帮助解决。理想状态下此DSS应当建立在合适的模型基础上。*合适的模型*简单而言指模型的复杂性不能太高不易操作及控制，也不能过于简单从而失去了科学合理性。

然而，DSS的设计仍然存在诸多困难，具体表现为:(1)缺乏切实可行的模型选择方法，即在满足其功能要求的同时得到适当的模型复杂性；(2)缺乏对DSS整体表现的评估，尤其是对系统中不同部分，如模型，参数等方面的不确定性的分析。这两个方面的工作是DSS设计的关键。这些困难在所有涉及多目标，多学科以及不同时间和空间范围的DSS设计中都有发现。

从设计角度而言，综合河流管理DSS的设计大都基于系统分析法。该方法从对河流物理条件的分析入手，定义管理目标，然后根据功能要求来构造和确定系模拟系统。论文对DSS发展历史上一些典型的设计方法进行了研究比较。研究发现在这些无论是以用户为本或从科学出发为原理的设计方法中，模型的复杂性确定没有得到应有的重视。比如，以用户为本的设计原理主要要考虑用户的要求，该方法可能因为用户对专业不够理解而不得不选择一些简单的模型，或者就地取材选用手中已有或者自己开发的模型软件，其结果可能导致模型过于简单而达不到预期效果。以科学为本的设计理念从則科学原理出发，模型的设计尽量满足科学原理的要求，务求设计出的DSS能被用到任何河流解决任何问题。这种设计理念的产物的确满足了系统的科学性和合理性，却也可能导致模型复杂庞大，不易被用户理解和接受，更不用提其他不必要的人工及模型建設花费,如此也达不到DSS务求*简单合理*的设计要求。总之，这些DSS设计方法中，“怎样合理选择模型的复杂性?”没有得到相應的重视和解决。

通常情况下，当需要从一些模型中挑选出某一个模型时，首先对他们的表现进行比较，然后對照考核標準选出表现最好的一种，這可能可以作為模型選擇的方法。但是，这种方法在DSS设计中很少出现。实际上，模型的选择都是在对事物及现象进行分析后有针对进行的，或者(大部分情况下)充分利用现有模型。对模型的表现进行比较与其说是选择模型的方式，不如说是科学研究行为来得妥当。模型的表现也因人而异，很大程度上依赖于模型使用者的专业知识及经验。另外，模型复杂性的选择的因具体要求而不同，比如单目标管理(如防洪)和多目标管理(如防洪与生态相结合)的模型复杂性是不一样的。这些都令到在实践中很难断定某一个/些 模型比另一个/些更好。

因此，要得到一个和适的DSS，有两个问题必须解决：(1)怎样确定和选择合适的模型复杂性？(2)如何对DSS的整体表现有一个合理的评估，以保证DSS的确能够且有效支持决策的确定？本论文對这两个问题进行了充份讨论和研究。首先,论文提出了一个以系统分析法为基础的DSS设计大纲。该大纲包括两个独立的设计过程：*质析*及*量解*。质析过程中对设计目标，所要解决的管理问题，流域物理条件等进分析，从而找到相应的学科和参数，通过这些学科及参数之间的驱动关系定出一个概念性的DSS系统结构；量解过程着重于模型的建立，模型率定及测试，以及不确定性分析等。對以上兩個針對性強的問題,論文也提出了解决途徑。

在质析过程中，為確定模型的複雜性,论文提出了*双向搜索法*。该方法的核心在于，在已确定的环境及社会经济指标的指导下，通过学科及参数之间的驱动关系来确定模型的复杂性。该方法包括两个相互指向的过程，及*前进式搜索*和*后推式搜索*。前进式搜索通过对问题和流域物理条件的分析来确定哪些物理学科（比如水文学，水力学或者生态学等)必须包括进来，后推式搜索则应用各种参数指标来约束模型复杂性的扩散。

對怎样评估此DSS是否合理有效,在众多方法中，不确定性分析在回答此问题上具有非常重要的作用。由于可以帮助系统建立者了解各不确定性在系统各部件之间的分布，以及得到DSS的综合表现，不确定性分析法可以提供决策支持更为全面的系统结果。但是，从技术上而言，DSS的不确定性分析仍然有一定难度。过去的不确定性分析大都是对单模型的分析，涉及多模型大量数学计算的系统如DSS的不确定性分析鲜有报导。该问题在此论文中也通过对*方案树法*(第二章)的应用而得以解决。

应用上述DSS设计步骤，结合德国ELBE河的洪水管理目标，本论文给出了一个综合洪水风险评估(IFRA)大纲。该大纲被用来对洪水的局地及非局地影响进行了风险评估，具體如下文所述。

在洪水风险方法发展过程中，有两种风险分析方法比较普遍：即*统计法*和*物理分析法*。基于比较简单的水位~流量关系，统计法计算年平均洪水毁损，此结果一般可用于实现长期风险管理目标比如设计和建筑防洪大坝，该方法主要适用于用于比较大的地理范围比如流域范围(几百至几千公里跨度),也可以用于比较小范围局地区域；物理分析法则计算次洪及流域物理变化所引起的的淹没损失。物理分析法一般采用二维水力学模型来模拟洪水淹没深度及流速等。该方法可以用来支持短期洪水管理行为比如分洪措施的实施等。由于大型数学计算的应用，物理分析法比较适用与较小的空间尺度，如100公里範圍內。

但是，目前洪水風險分析的不足之處在於：(1)淹沒損失沒有計入流速所造成的影響；(2)目前的淹沒計算主要基於二維水力學模型數學,該淹沒計算需要大量的數據支持,同時也需要大量計算時間,且不談模型建立及率定所需工作量。這些不足嚴重影響了淹沒計算模擬的靈活性和其實用性。

為將流速損失量化入洪水風險計算中,該論文提出了風險矩陣的方法。通過風險矩阵的使用,流速影響以分級方式與淹沒損失相結合,从而得到四個不同的風險水平。這些風險水平在實踐中與洪水管理行為相互對應。風險水平的使用雖然沒有直接將流速影響量化,但是在一定程度上考慮了流速造成的額外損失。風險矩陣法在第三章中有詳細描述。

對于第二個問題,由于遙感技術的提高以及地理信息系統(GIS)的廣泛應用及快速發展,快速洪水風險分析评估可以從這方面尋求解決途徑。本本論文中應用GIS技術,結合一維擴散波河流演算,對淹沒洪水進行了近似计算，并将此方法的計算結果與二維完全動力演算模型SOBEK1D2D的計算結果進行了比較。

應用上述方法,為測試其適用性,該使用IFRA对德國ELBE河SANDAU附近進行了兩個不同的洪水案例研究:即局地(第四章)和非局地(第五章)洪水風險及其影響。局地洪水風險分析主要分析洪災發生時洪水對受災地區的影響,非局地分析研究風險轉移措施在洪水中的應用效果,如破堤分洪措施。

在局地分析中,使用不同的洪水風險分析方法,論文中對兩個不同假定案例進行了分析比較,即,(1)使用統計法對有無堤防設施進行比較;(2)使用物理分析法研究破隄對防洪的影響。計算結果顯示了不同情況下洪水風險指標的明顯區別。不確定性分析的應用使得比較結果更为清楚明白, 因此,該系統可以用來支持決策的決定。

在非局地分析中,對破堤分洪措施的影響通過二維物理分析法進行了研究。結果表明嚴格控制下的意向性破堤分洪可以有效降低下游洪水風險及淹沒損失。在非局地分析中,不確定性分析技術還被用來研究基於動力模型的淹沒模擬中涉及的主不確定性,主要為數字地形圖(DEM)和糙率。研究結果表明不同的GIS數據處理方法(主要指為降低地圖分辨率以便模型使用)对淹没損失帶來很大的不確定性。而河流糙率對于流速不確定性貢獻較大,對于以淹沒深度來計算的洪災損失貢獻则不明显。此結果指出了流速對洪災損失計算不確定性的影響之嚴重:洪水流速對土地應用的敏感反應反過來證明了不同的土地使用對流速的敏感反應,比如房屋可能被摧毀等。

在非局地不確定性分析中的技術問題,即怎樣得到代表性足夠又不需要大量計算機時間上,*案例樹法*的應用的確效果显著。該法可被應用于大型數學模擬運算的不確定性分析。

研究同時發現,在洪水風險評估中,水力學模型的選擇取決于洪水風險管理目標。对于快速洪水风险评估，文中推出的GIS技術與一維水力學模擬相結合可以達到良好的淹沒計算,淹沒深度及面積結果與二維完全動力模型SOBEK1D2D計算結果非常吻合。但是,沒有動力演算而僅僅靠GIS技術來近似計算流速并不理想。这也許可以作为將來水力學演算的研究方向和內容之一。

About the Author

Ms. Yan Huang was born in a mountain village in Guizhou, Southwest China. Her dream, ever since she knew the meaning of outside, was to see the world with her own eyes.

In 1992, after graduating from Hohai University in Nanjing, East China, in the field of hydrology, she was offered a job at the Yangtze River Water Resources Commission in Wuhan, Central China. There she has been working on flood forecasting for the Yangtze River for four years. In 1996, she started working at the Danish Hydraulic Institute (DHI) in Denmark. Her international experience was further extended when, in October 1997, she participated the Hydroinformatics MSc programme at UNESCO-IHE Delft, the Netherlands. Later in 1998 she carried out her MSc thesis research on 'data assimilation for flood forecasting using data-driven methods' in DHI. She further applied the data-driven approach to the simulation of algal dynamics in the coastal waters of Hong Kong from 1999 to 2001, for which she was rewarded the degree of Master of Philosophy (MPhil) from Hong Kong University.

In September 2001, she returned to the Netherlands to start her PhD research. The research was partially funded by the German Ministry of Education and Science (BMBF) under the coordination of the German Federal Institute of Hydrology (BFG), Germany. The project aimed to develop an integrated decision-support system for management of the Elbe River in Germany. This provided an opportunity for carrying out research on developing an integrated flood risk assessment system incorporating uncertainty analysis and building on GIS technology. The Elbe project provided this thesis with an excellent case study. From April-June 2004 she visited Cornell University, USA, to carry out her research particularly on uncertainty analysis of hydrodynamic modeling. Upon return to the Netherlands, she worked on dynamic flood simulation modeling and accomplished her PhD in October 2005.

She intends to return to China to advance her professional career.