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Book of Abstracts
Bridging gaps between river science, governance and management

NCR-days 2015
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H.J.R. Lenders, F.P.L. Collas, G.W. Geerling and R.S.E.W. Leuven (eds.)

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NCR-Days 2015

The Netherlands Centre for River Studies (NCR) has been established on October 8th 1998. It is a collaboration of nine Dutch research institutes with the goal to enhance cooperation in the field of river-related research. An important activity of NCR is to organize the NCR-days, an annual conference organized in rotation by the institute members. The 2015 edition of the NCR-days is organized in Nijmegen by the Institute of Water and Wetland Research (IWWR) and the Institute for Science, Innovation and Society (ISIS) of Radboud University.

We invited six international renowned scientists/practitioners to elaborate on the key question of the NCR-days 2015: How to bridge gaps between river science, governance and management? **Prof. Hans de Kroon**, director of the IWWR (Radboud University), gives the opening speech. He discusses how past, current and future IWWR research may contribute to river research and river management. **Dr. Tom Buijse** (Deltares) illustrates how the interdisciplinary EU REFORM programme contributes to tools for cost-effective restoration of river ecosystems and improved monitoring of the biological effects of physical change by investigating natural, degradation and restoration processes in a wide range of river types across Europe. **Prof. Hervé Piegay** (University of Lyon) elucidates feedbacks on interdisciplinary research in riverine sciences, based on 35 years of experiences and involvement of four generations of river scientists in Eastern France. **Prof. Maarten Kleinhans** (Utrecht University) presents novel experimental and modelling approaches to unravel biomorphodynamics of river estuaries and to predict past and future, large-scale effects of climate change. **Prof. Suzanne Hulscher** (University of Twente) discusses progress in RiverCare. This large research programme of NCR partners aims at a better understanding of fundamental processes that drive ecomorphological changes in rivers, predict the intermediate and long-term developments and develop best practices to reduce the maintenance costs and increase the benefits of interventions. It is funded within the Perspective Programme of the Dutch Science and Technology Foundation (STW) and supported by many public and private partners. The keynote of **Eric Schellekens MSc** from ARCADIS focuses on the implementation of fundamental and applied river science in sustainable management measures, while using high level solutions, approaches for adaptive stakeholder participation, frontrunner knowledge and expertise, and experiences beyond the core ‘technical’ scope. He provides food for thought on bridging the gaps between river science, governance and management in the international water market.

The usual sessions consist of 20 oral presentations and 13 poster pitches. These contributions are a sample of the current river research and focus on ecological, hydrological, morphological and socio-economic studies, including theoretical, experimental and (integrative) modelling approaches. We hope to offer two inspiring NCR-days with a programme that yields much discussion, novel ideas and new opportunities for cooperation. We would like to thank the Behavioural Science Institute (Radboud University) for providing the poster displays, all members of the NCR programme committee, Koen Berends (secretary of the NCR programme committee), Monique te Vaarwerk (secretary of the NCR, University of Twente), Lizette Donders and Vera Janssen (Institute for Science, Innovation and Society, Radboud University), Gina Delmee and Marianne Uijt de Haag (Department of Environmental Science), and our co-organisers Jan Fliervoet, Remon Koopman, Swinda Pfau and Laura Verbrugge for their immense support during the organisation of these NCR Days. Last but not least, the financial support of the Netherlands Organisation for Scientific Research (NWO) is greatly acknowledged.

Rob Lenders, Frank Collas, Gertjan Geerling and Rob Leuven

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1 – Presentations day 1

The Institute for Water and Wetland Research: ecological research as a fundament for innovative solutions at the boundary between land and water

H. de Kroon[1]
[1] *Department of Plant Ecology, Institute for Water and Wetland Research, Radboud University, Heyendaalseweg 135, 6010 GL Nijmegen, the Netherlands.*
Email: H.deKroon@science.ru.nl

Abstract
The ecologists, physiologists, microbiologists and environmental scientists of Radboud University have organized themselves in the Institute for Water and Wetland Research (IWWR). It is the aim of the IWWR is to increase our understanding of environmental stress responses of wetland systems at various levels of organization; from cellular levels via organisms to ecosystems. In particular we aim to elucidate the adaptation of micro-organisms, plants and animals to changes in water quantity and quality under aerobic or anaerobic conditions.

The IWWR is unique by working on three major groups of organisms (micro-organisms, plants, animals) as well as by working at a large variety of organization levels, from genetic and genomic studies at the molecular level up to global models on the impact of environmental stressors. Together, the IWWR coherently covers stress responses to water problems from molecular mechanisms to changes at the global scale.

The mission of IWWR is not only to perform ground-breaking research on adaptation to environmental stress, but also to contribute to innovative solutions of pressing environmental problems related to flooding, drought and water pollution. We realize this aim in close and often long-standing interactions with companies, consultancies, and governmental and non-governmental organizations. These partners directly profit from discoveries and novel insights obtained in IWWR fundamental research. Examples include novel waste water treatment technology, environmental impact assessments, ecological risk analyses of chemical and physical stressors, numerous successful wetland restoration projects, fish welfare improvement, wild species management recommendations, strategic plant breeding research and patent applications. Vice-versa, societal challenges are a source of inspiration for new fundamental research. In this way, the intimate relationships between IWWR thematic research and partners results in a fruitful cross-fertilization between top fundamental research and innovative applications. Two of our partners on campus, the company B-Ware (Biogeochemical Water-management & Applied Research on Ecosystems), and the NGO's organised in 'Natuurplaza', play a special role in this respect.

I will give a number of examples of how past, current and future IWWR research may contribute to river research and river management. For example, the flooding tolerance of riverine plant species determines the species distribution in floodplains, the composition of plant communities, and the erosion and sedimentation properties of their sites. Discoveries on the rooting behaviour of species in communities of different composition may help to design dike vegetation that can better cope with flood erosion and drought, is more attractive and cheaper in maintenance. The IWWR obtained an ERDF grant (European Fund for Regional Development) in 2012 to realize an integral system in which all important wetland ecosystem services (water retention, generation of energy, water purification, mineral recycling) are developed together with local and regional authorities and a variety of companies. Recent work of IWWR population ecologists together with Sovon (Dutch Centre for Field Ornithology; partner in 'Natuurplaza') highlighted the potentially devastating effects of low concentrations of neonicotinoid insecticides in the surface water on bird populations, raising pertinent questions about sources of pollution and waste water purification. Finally, IWWR together with 'Natuurplaza' is host to the Netherlands Centre of Expertise on Exotic species (NEC-E) that also performs fundamental and applied research in riverine areas. Invasive exotic species strongly dominate biodiversity and functioning of riverine ecosystems.

Hydromorphology of rivers and floodplains – what is at stake and how does REFORM contribute?

A.D. Buijse[1], E. Mosselman[1], I.G. Cowx[2], N. Friberg[3], A.M. Gurnell[4], D. Hering[5], E. Kampa[6], C. Wolter[7]
[1] *Deltares, DELFT, the Netherlands*
[2] *University of Hull, HULL, UK*
[3] *NIVA, OSLO, Norway*
[4] *Queen Mary University of London, LONDON, UK*
[5] *University of Duisburg-Essen, ESSEN, Germany*
[6] *Ecologic Institute, BERLIN, Germany*
[7] *Leibnitz-Institute of Freshwater Ecology and Inland Fisheries, BERLIN, Germany*
Email: tom.buijse@deltares.nl

Context and objectives

Europe is characterized by a dense network of rivers that provide essential ecosystem services. From an ecological perspective, rivers and their floodplains form some of the most diverse ecosystems worldwide. Recent analysis of the first round of WFD River Basin Management Plans (RBMP) indicated that 40% of European rivers are affected by hydromorphological (HYMO) pressures caused predominantly by hydropower, navigation, agriculture, flood protection and urban development. As a consequence, there is increasing emphasis on river restoration driven by demands of the WFD and EU States have drafted programmes of measures focusing on restoring river hydrology and morphology. Implementation will require substantial investment in these measures, but there remains a great need to better understand and predict the costs and benefits of future river restoration.

Against this background, REFORM's goal is to generate tools for cost-effective restoration of river ecosystems, and for improved monitoring of the biological effects of physical change by investigating natural, degradation and restoration processes in a wide range of river types across Europe. The consortium is composed of 26 partners from 15 European countries representing a wide range of disciplines: hydrology; hydraulics; geomorphology; ecology; socio-economics; and water management. REFORM's objectives are grouped into three categories: application, research and dissemination.

Application

1. Select indicators for cost-effective monitoring of physical habitat degradation and restoration.
2. Improve tools and guidelines for HYMO restoration and mitigation.

Research

3. Review existing information on river degradation and restoration.
4. Develop a process-based HYMO framework relevant for ecology and suitable for monitoring.
5. Understand how HYMO pressures interact with other stressors and constrain restoration.
6. Assess the importance of scaling on the effectiveness of restoration.
7. Develop instruments for risk and benefit analysis to support successful restoration.

Dissemination

8. To increase awareness and appreciation for the need, potential and benefits of river restoration through active interaction with stakeholders.

Interim results

The REFORM project has generated substantial mid-term outputs to support River Basin Management Planning for the Water Framework Directive.

Interim results have been synthesised and made available to practitioners in an accessible way by the set-up and population of a WIKI (http://wiki.reformrivers.eu; Mosselman et al. 2013).

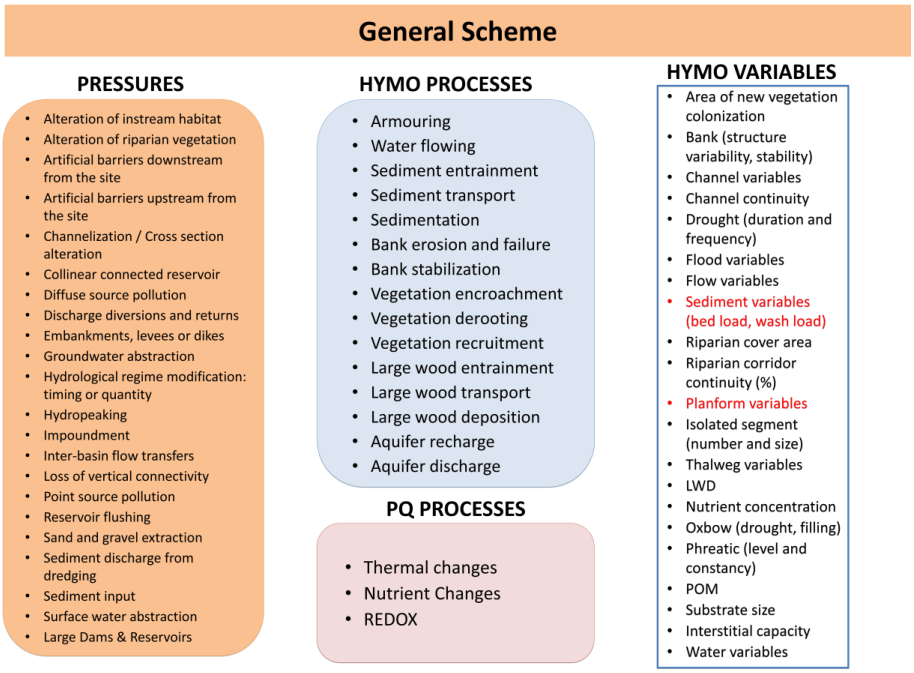

Figure 1. Hydromorphological pressures that have been reviewed for their impact on HYMO processes.

Key HYMO processes and variables indicating. success in river restoration have been reviewed (Garcia de Jalon et al. 2013; Figure 1. HYMO variables that influence ecological status and functioning have been linked to the tolerance thresholds of species with emphasis on macrophytes, macroinvertebrates and fish (Wolter et al. 2013). The dynamics of flowing water emerged as the most important HYMO process. Coarse gravel maintained by stream power and flow velocity emerged as key indicator. Significant knowledge gaps need to be addressed for habitat requirements of riverine species.

A review of case studies and literature on costs and benefits of river restoration in Europe showed that cost data are quite variable and usually not available in a form appropriate for further assessments (Ayres et al. 2014). Thus, investing efforts in standards and protocols to gather and incorporate cost information in a more systematic way will benefit decision-making.

In assessing hydromorphology to date there has been too strong a reliance on the reach scale. For sustainable solutions, it is crucial to develop understanding of the functioning of river reaches in the wider spatial context. The ways in which river reaches have responded to changes in the past, provide crucial information for forecasting how reaches may change in the future. The REFORM framework allows users to incorporate all of these multi-scale spatial and temporal aspects into river assessment and management (Gurnell et al. 2014a).

Riparian vegetation is not included as a biological quality element in the Water Framework Directive. Gurnell et al. (2014b) present new science concepts and analyses that clearly demonstrate the importance of riparian and aquatic vegetation as a key physical control on river form and dynamics and a crucial component of river restoration.

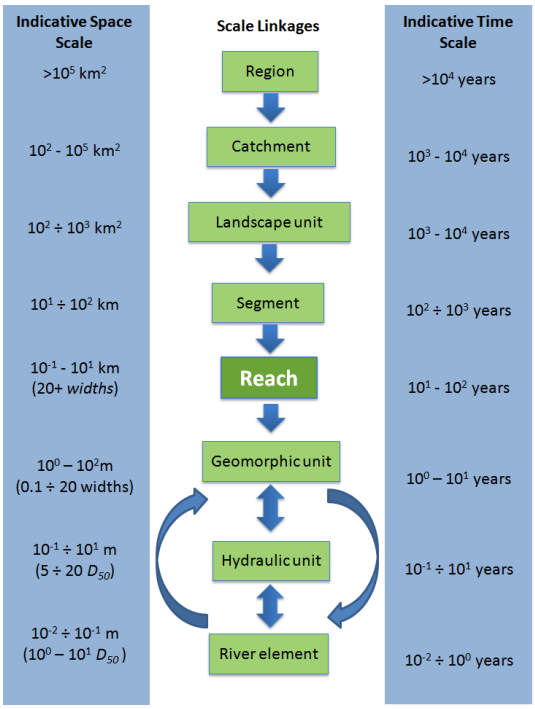

Figure 2. Hierarchy of spatial scales for the European Framework for hydromorphology, including indicative spatial dimensions and timescales over which these units are likely to persist.

Existing metrics have been evaluated for their strength to distinguish the impact of HYMO pressure on the mandatory biological quality elements from other stressors (Friberg et al. 2013). This showed that there is potential to develop metrics from monitoring data on fish and macrophytes to indicate HYMO impacts. Contrarily, relationships between HYMO

degradation and macroinvertebrate metrics were weak.

Despite the rapid increase in river restoration projects, little is known about the effectiveness of these efforts and many practitioners do not follow a systematic approach for planning restoration projects. REFORM has developed a planning protocol that incorporates benchmarking and setting specific and measurable targets for restoration and mitigation measures (Cowx et al. 2013).

Existing data on the effect of restoration on biota complemented by information on factors which potentially enhance or constrain this were analysed (Kail & Angelopoulos 2014). Overall, restoration success did most strongly depend on project age and river width, and was affected by agricultural land use. Restoration still had a positive effect in catchments dominated by agricultural land use, and thus do not question the implementation of restoration projects in intensively used catchments. The influence of project age stresses the need for long-time monitoring to investigate the restoration effect over time.

Expected final results and their potential impact and use

From analysis of the first round of RBMPs, it is clear that the hydrology in many river basins and the morphology of streams, rivers, riparian zones and floodplains have been modified to serve social and economic needs (flood protection, water supply for agriculture, households and industries, navigation, hydropower). The ecological impact is poorly understood. To remediate this EU Member States have set up programmes of measures to improve the ecological status of their water bodies. REFORM is the first European research project to have a strong emphasis on supporting the knowledge base for the programmes of measures, i.e. how to restore our rivers. Reflecting the foreseen outcome of REFORM with the main topics raised during its stakeholder workshop in February 2013 clearly highlighted the potential use of its results. REFORM has contributed by improving our scientific understanding of the linkages between hydromorphology and ecological status, but moreover by making its results available in various forms to support both practitioners and scientists.

Besides expanding the knowledge base, there is also an urgent need to share experiences. Web-based knowledge information systems are an effective means to share the know-how from practical experience and connect this with the scientific knowledge. Consequently REFORM has developed a WIKI that has been populated throughout the course of the project with information relevant for various phases of River Basin Management Planning (characterisation of basins and water bodies, objectives, impacts of hydromorphological pressures, programmes of restoration measures) to meet this need (Figure 3). Complementary, the LIFE+ project RESTORE has generated the largest European database on stream and river restoration projects (https://restorerivers.eu/wiki).

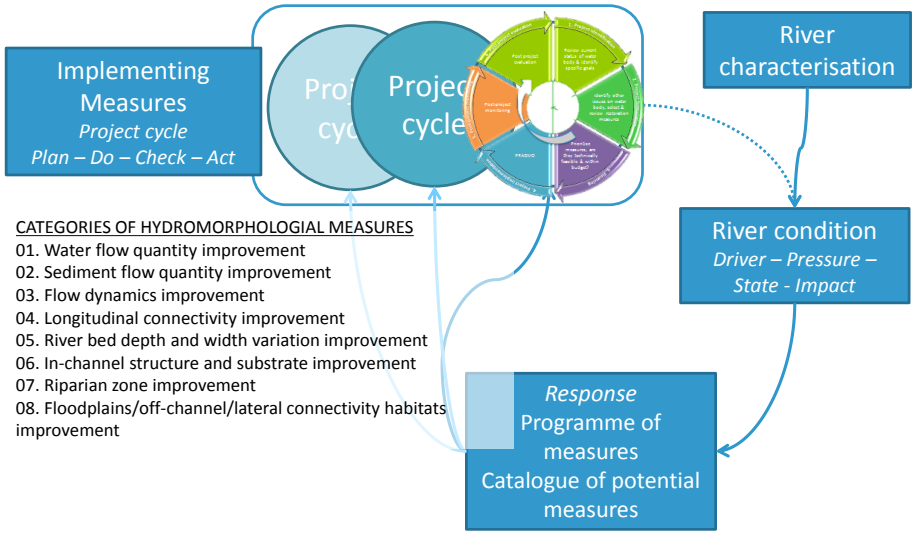

Figure 3. In the online WIKI the results of REFORM's are presented in the implementation cycle of River Basin Management Plans.

Practitioners will benefit from the tools for improved monitoring of the impact of HYMO pressures and for better planning and evaluating of restoration. They will also benefit from the WIKI, which is a more effective way to trace relevant information on these topics than currently available. On the other hand, scientists benefit from a wide range of scientific publications regarding the role of scale and processes to shape rivers and streams by hydromorphology and vegetation, to discern the impact of hydromorphological pressures on biota from other stressors, the extent to which scale matters for restoration, and answers to

the question how restoration could become more effective in all phases of project realisation cycles.

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The influence of the upper Kapuas wetland area on the Kapuas flow characteristics, West-Kalimantan, Indonesia

D.C.C. Bol
Droevendaalsesteeg 67, 6708 PN Wageningen,
Email: dinja.bol@gmail.com

Introduction

The Kapuas, located in West-Kalimantan, Indonesia, is a hydrologically interesting system. It has a seasonally flooded inland wetland, which alternately feeds and drains the Kapuas river (Klepper et al., 1996). This wetland is often believed to have a regulating role on the river discharges. In order to investigate the role of the lakes, a model study is performed.

Methods

To investigate the hydrology two modelling approaches are used. A simplified water balance model of the entire lumped catchment is used to determine the general discharge behaviour. As second approach the simplified water balance model is used to determine the incoming lateral flux to the river for 49 sub catchments. In series these lateral inputs are coupled to a 1D river flow model.

The period Oct-2013 to Sep-2014 is modelled. The catchment area consists of the entire water basin of the Kapuas in West-Kalimantan until Sanggau, 553km upstream. For the hydrological assessment, the relatively new 3B42V7 product of the Multi-Satellite Precipitation Analysis (TMPA) Tropical Rainfall Measuring Mission (TRMM) is used. This product seems to be promising for hydrological purposes.

The simplified water balance model is created in R and distributes the incoming water over three flow routes: 1 % surface runoff, 26% soil quick flow and 73% base flow. The fluxes are calculated in parallel according to the leaky bucket concept:

$$Q_i = Q_{i-1} + (P_i - Q_{i-1}) / K \qquad (1)$$

The discharge (Q) at day (i) is dependent on the discharge on the previous day (Q_{i-1}), the incoming effective precipitation (P) and a reservoir constant (K). The reservoir constant determines the outflow and has a value of 1, 5 and 20 days for the three flow routes respectively.

From the elevation differences, as measured by SRTM, the Shuttle Radar Topography Mission, 49 sub catchments were created which discharge into the Kapuas river.

The 1D representation of the Kapuas river is modelled in the River Analysis System HEC-RAS (U.S Army Corps of Engineers, 2002). The Saint-Venant equations are solved on a 20min and 3000m resolution. Next to the Kapuas river itself, the catchment is characterized by a large wetland area, which can act as a buffer on the river flows. This wetland area is modelled as one large reservoir lake, which connects to the Kapuas river via the Tawang channel, 354km upstream of Sanggau. The channel alternately feeds and drains the Kapuas river depending on the river and lake stages.

Results

The rainfall pattern shows two distinct wet seasons, a major one in November, December and a minor one in March, April. The discharge at Sanggau displays an equal behaviour, with a delay in time of about a month.

The lumped water balance model simulates the discharge at Sanggau fairly well, with a Nash-Sutcliffe Efficency of 0.78. The simulation displays a too peaky behaviour on a weekly timescale and the falling limb of the hydrograph is too steep in the first dry season.

In HEC-RAS the river dynamics over the entire river reach are simulated. During the rising limb of the hydrograph a backwater curve arises just upstream of the Tawang channel. During the falling limb, a smooth surface water profile develops.

A scenario with and without wetland is performed to assess the influence on the river discharges. Upstream of the Tawang channel the difference between these scenarios is negligible. Directly downstream of the Tawang channel, the river discharge can consist for 60% of lake water fed to the Kapuas. The wetland leads to a more gradual discharge behaviour and a 17 day shift in time of the peak flows. Next to the discharges the flow velocities are affected. The backwater curve which arises upstream of the Tawang channel, causes the flow to alternately increase and decrease in velocity, while without wetland a constant decrease could be seen.

The effect on the river discharges diminishes while moving downstream, as more water adds laterally to the river. At

Sanggau the shift in time vanished. The discharges during the rising limb are approximately equal, only during the falling limb an effect of the wetland can be seen, figure 1. The falling limb occurs more gradually and the minimum flow changes from 929 to 1038 m^3/s.

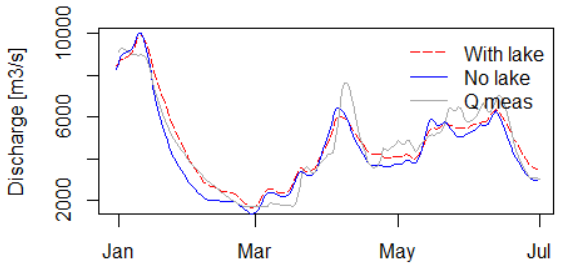

Figure 1. Discharge simulations at Sanggau with lake and no lake scenario in HEC-RAS.

Conclusion

The alternately feeding and draining Kapuas wetland causes a shift in time in the peak flows directly downstream of the Tawang channel. As the river discharge at Sanggau consist for 60% of water which adds to the river after this channel, during the rising limb of the hydrograph the effect of the wetland at Sanggau is negligible. During the falling limb the wetland provides an extra supply of water.

The changing flow velocities upstream of the Tawang channel might explain the excessive meandering behaviour of the Upper Kapuas. The deviation in measured and simulated discharge amount might be caused by the TRMM measurements. During the second wet season, precipitation fell mainly at the eastern boundary, which is characterized by a large elevation difference. TRMM is shown to be less accurate in mountainous catchment (Li et al, 2012), which might cause the deviation in April, May.

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Potential damping of extreme floods in the river Meuse between Ampsin (B) and the Belgian-Dutch border

M. Bruwier, P. Archambeau, S. Erpicum, M. Pirotton, B. Dewals
Hydraulics in Environmental and Civil Engineering (HECE), University of Liege (ULG), Liege, Belgium
Email: mbruwier@ulg.ac.be

Introduction

In the framework of the Interreg IVB project AMICE, hydraulic modelling of floods in the river Meuse was performed by coordinating existing models (Dewals et al. 2012a). The influence of climate change scenarios was incorporated indirectly in the simulations through a transnational hydrological scenario (Drogue et al. 2010; Dewals et al. 2013). For a "wet" future climate, this scenario assumes that the peak discharge Q_{100} of the 100-year flood would increase by 15 % for the time horizon 2021-2050 and by 30 % for the time horizon 2071-2100.

However, the design discharge currently used for planning in the Netherlands is higher than those considered to date in the AMICE project. From this perspective, it becomes relevant to analyse flood scenarios corresponding to a peak discharge above Q_{100}+30 %, referred to hereafter as an "extreme" flood.

The aim of the study is to give an appreciation of the potential damping of one extreme flood scenario along the river Meuse between Ampsin (Belgium) and the Belgian-Dutch border (Figure 1). This analysis is based on the hydraulic model WOLF 2D (Erpicum et al. 2010) applied to a coarse grid with simplifications in the schematisation. The hydraulic model WOLF 2D has been already used in several studies to simulate floods (Ernst et al. 2010; Beckers et al. 2013, Bruwier et al. 2015, Detrembleur et al. 2015). The coarsening of the grid for simulating the extreme flood scenario is necessary to preserve the computational efficiency, since both the total flood duration and the inundation extents (hence the number of grid cells) increase considerably between the Q_{100} + 30 % scenario considered previously and the extreme scenario considered here.

The main steps presented in this abstract are the following:

- presentation of a method for the coarsening of the topographic data;
- definition of the flood waves to be considered;
- calibration/validation of the model based on the results of AMICE;
- simulation run for a flood wave with a peak discharge close to 5000 m³/s at the Belgian-Dutch border;
- analysis and interpretation of the results in terms of amount of flood damping;
- conclusions and perspectives.

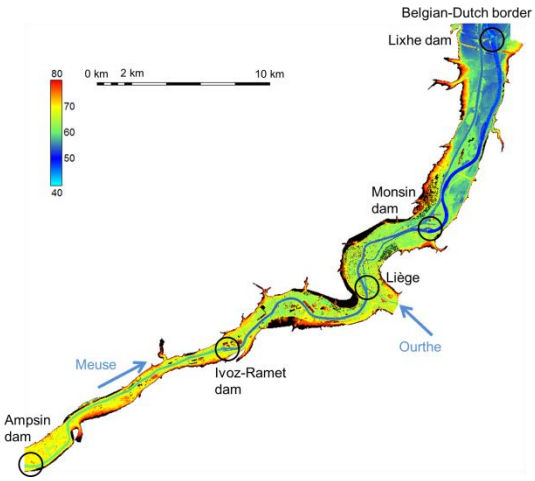

Figure 1. River Meuse between Ampsin and the Belgian-Dutch border.

Method

Coarsening of the topographic data

From the previous hydraulic analyses conducted within the AMICE project, a detailed topographic model is readily available on a grid of 5 m by 5 m; but adaptation to a coarser mesh is needed here. In order to increase the computational efficiency, the size of the grid cells were multiplied by a factor 4: from 5 m to 20 m, allowing a potential reduction of the computational time by a factor 4^3 = 64. However, a simple grid coarsening would also decrease the accuracy of the results due to the loss of detailed topographic information. Nonetheless, two degrees of freedom can be used to minimize the degradation of the results due to the coarsening: the calibration of the roughness coefficient (Manning coefficient) and the coarsening method itself.

In this study, we propose to analyse the effect of a spatial distribution of the mathematical operation used to aggregate the fine scale data. This coarsening method is refered to as "spatially distributed aggregation". The goal of

a spatially distributed aggregation is to assign a suitable mathematical aggregation operation to each coarse cell as a function of the expected role of this coarse cell in the flow process.

Based on a fine scale map of water depth (5 m x 5 m) computed for a near-bankfull discharge (*main riverbed map* in Figure 2a), three classes of coarse cells are identified:

- if a coarse cell is filled by water with a wet area exceeding a threshold percentage *TP* (calibrated to *TP* = 25%) of its total area, this coarse cell is tagged as a *riverbed cell* (Figure 2b);
- a *riverbank cell* is a dry coarse cell adjacent to a riverbed cell (Figure 2c);
- a *floodplain cell* is a dry coarse cell not in direct contact with a riverbed cell (Figure 2c).

The mathematical aggregation operation used in each coarse cell is selected depending on the classification of the coarse cell:

- For a coarse riverbed cell, the mathematical aggregation is an average operation among the fine cells which (i) are included in the coarse riverbed cell and (ii) are identified as wet in the main riverbed map (to avoid taking into account riverbank levels);
- For a coarse riverbank cell, the aggregated value is the maximum value of the fine cells included in the coarse riverbank cell or in its surrounding coarse riverbed cell(s) (in order not to miss the maximum riverbank level).
- For a coarse floodplain cell, an average operation is performed for the *N* (calibrated to *N* = 14) smallest values attributed to the fine cells composing the coarse one in order to limit the influence of high levels of buildings on the aggregated value.

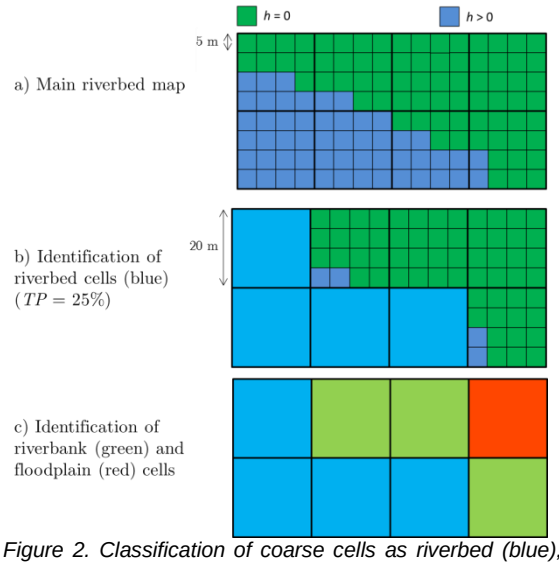

Figure 2. Classification of coarse cells as riverbed (blue), riverbank (green) or floodplain (red) cells from the main river map.

Definition of the flood waves

For conducting the unsteady simulation corresponding to the extreme flood scenario, a hydrograph was prescribed as upstream boundary condition at the upper border in Ampsin and at the junction with river Ourthe (Figure 1). The synthetic flood waves were determined following the same procedure as in the AMICE project (Dewals et al. 2012a; Dewals et al. 2012b) and were rescaled to reach the peak discharge corresponding to the considered extreme flood scenario (Figure 3). The time shift between the flood waves in the Meuse and in the Ourthe was determined so that the two peak discharges arrive approximately simultaneously at the junction between the Meuse and the Ourthe (Liège).

To start unsteady modelling, we considered as initial condition a steady-state corresponding to a near-bankfull discharge.

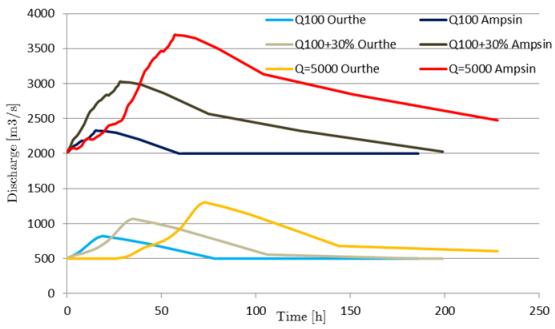

Figure 3. Hydrographs corresponding to the different flood scenarios.

The downstream boundary condition is a stage-discharge relationship obtained from Dutch modelling results at the border between Belgium and the Netherlands.

Calibration of the Manning coefficient of the coarse model

The Manning coefficient for the coarser model was calibrated for an unsteady calculation with a peak discharge Q_{100}+30% by comparison with the previous hydraulic analyses conducted within the AMICE project in order to reproduce with a good accuracy the damping of the flood discharge.

Results and discussion

Calibration and validation of the coarse model

For the values of the Manning coefficient given in Table 1, the damping of the flood discharge Q_{100}+30% is compared to the damping obtained with the fine grid simulations

(Figure 4). The damping is defined as the ratio between the computed peak discharge in the unsteady modelling and the sum of the peak discharges of the flood waves prescribed as boundary conditions at Ampsin and for the Ourthe. For configurations III and V, the error on the damping of the flood discharge is lower than 0.75% over the entire reach and lower than 0.30% at the Lixhe dam, 1 km at the upstream of the Belgian-Dutch border.

Table 1. Configurations used for unsteady calculations with a peak discharge of Q_{100} + 30%.

Reach	n ($x10^{-3}$ $s.m^{-1/3}$)				
	I	II	III	IV	V
Ampsin to Ivoz-Ramet	20	18	16	18	16
Ivoz-Ramet to Liège	20	18	16	18	16
Liège to Monsin	23	23	23	22	22
Monsin to the Belgian-Dutch border	23	23	23	22	22

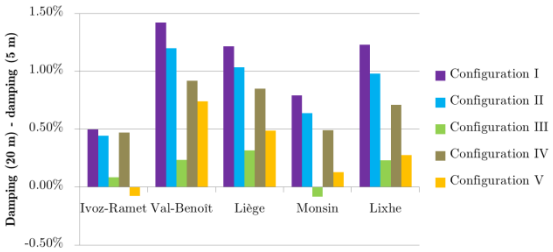

Figure 4. Difference between the damping of the flood discharge computed for the different coarse configurations and the peak discharges computed with the fine simulation.

Simulation run for a flood wave with a peak discharge close to 5000 m³/s at the Belgian-Dutch border

The two configurations (III and V) identified as those giving the best results for an unsteady calculation with Q_{100}+30% were used to simulate the extreme flood scenario with a peak discharge close to 5000 m^3/s at the Belgian-Dutch border.

The damping of the flood discharges computed for Q_{100}+30% in the AMICE project and computed for the extreme flood scenario with configurations III and V are compared in Figure 5. From Liège to the border between Belgium and the Netherlands, the damping is lower for configuration V than for configuration III, consistently with the reduction of flood levels (smaller Manning coefficient) in the two downstream reaches in configuration V. The differences in damping between configuration III and V are lower than 0.5% for the entire river chainage and around 0.2% at the border between Belgium and The Netherlands. At the border, the damping of the extreme flood discharge is evaluated around 3%. Over the entire chainage, the damping of the extreme flood discharge is lower than the damping computed for Q_{100}+30% in the AMICE project.

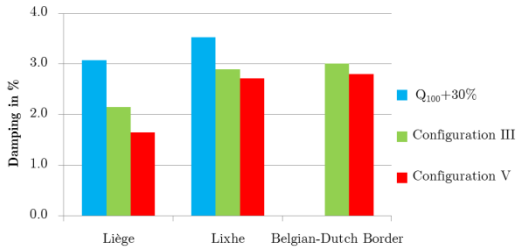

Figure 5. Damping of the peak discharge for Q_{100}+30% and for the extreme flood scenario with configurations III and VI.

As shown in Table 2, the volume of water stored in the floodplains increases by 30 hm^3 (+263%) between Q_{100}+30% and the extreme flood discharge. Considering that the AMICE results revealed significant inundations for Q_{100}+30% (~4100 m^3/s at the Belgian-Dutch border) but not for Q_{100}+15% (~3600 m^3/s), we can reasonably assume that the first significant inundations occur for discharges slightly above 3600 m^3/s. The volumes of water brought by the flood waves for discharges above 3600 m^3/s are respectively about 43 hm^3 and about 248 hm^3 for Q_{100}+30% and for the extreme flood scenario (Figure 6). In the Q_{100}+30%, the volume stored in the floodplains (11 hm^3) represents about 25% of the volume conveyed by the flood wave above the assumed threshold for inundation (43 hm^3). In contrast, for the extreme flood scenario, this ration drops to about 16% (40 hm^3/248 hm^3). Consequently, the storage capacity of the floodplains for the extreme discharge is reached well before the peak discharge is reached, leading thus to a relatively low damping.

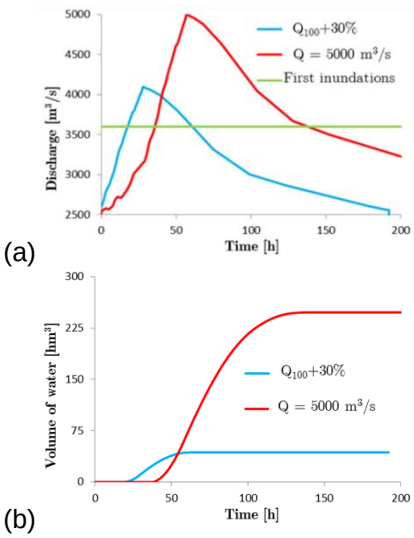

Figure 6. Input hydrographs (a) and cumulative volume of water above a discharge of 3600 m^3/s (b) for Q_{100}+30% and the extreme flood scenario.

Table 2. Comparison of maximum inundation extents and stored volumes for Q_{100}+30% and the extreme flood scenario.

	Q_{100}+30%	Q = 5000 m^3/s
Inundation extent	2000 ha	3200 ha
Stored volume	11 hm^3	40 hm^3

Conclusion

The goal of this study is the evaluation of the potential damping of an extreme flood wave in the river Meuse between Ampsin (Belgium) and the Belgian-Dutch border. The considered flood scenario corresponds to a peak discharge close to 5000 m^3/s at the Belgian-Dutch border. The study capitalizes upon the knowledge gained previously during the AMICE project, in which discharges up to 4095 m^3/s (Q_{100}+30%) were simulated.

The grid spacing used during the AMICE project (5 m) has been coarsened (20 m) to reduce the computational time. To preserve the accuracy of the model, topographic data have been aggregated following an original spatially distributed approach taking into account a classification of the computational cells as *riverbed*, *riverbank* or *floodplain* cells. The friction coefficient has been calibrated through comparisons of the computed results with unsteady results from the AMICE project. For Q_{100}+30%, the coarse model was shown to predict the damping of the peak discharge (3.5%) with an absolute error of the order of 0.5%.

The damping of the extreme flood discharge with a peak discharge close to 5000 m^3/s has been estimated close to 3%. The low value of the damping rate is a consequence of the relatively small volume of water stored in the floodplains compared to the volume brought by the flood wave for discharges levels leading to inundations. Nonetheless, a sensitivity analysis of the results of the present study with respect to the shape of the flood wave would contribute to further elucidate the influence of this shape on the rates of flood damping.

This study has also led to the development of an efficient hydraulic model to simulate extreme floods in the river Meuse from Ampsin of the Belgian-Dutch border.

Acknowledgements

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One-dimensional lake models with wind effects

K. Horváth
Deltares, Department of River dynamics and inland water transport Boussinesqweg 1, 2629 HV Delft

Introduction

One dimensional models are an efficient way of calculating water levels in rivers, when modelling river systems. However, often there are two dimensional water bodies connected to river systems, like lakes. In order to have one general model of the system these water bodies, having a rather two dimensional character, should also be modelled in a one dimensional environment.
In case of shallow lakes, the effect of the wind is one of the most influential factors determining the water level. The wind coming from different directions can be modelled in two dimensions quite naturally, however, a simple one dimensional model does not capture this influence by default.

Motivation of the research

This study is about the setup of a modelling approach for shallow lakes which should correctly represent the effect of the wind with a one dimensional hydraulic model.

This document is structured as follows: first the theoretical background and the tools are presented, then some partial results of the on-going study are shown.

Background

One dimensional models capture the effect of the wind by modelling the shear stress that effects the water level in the direction parallel to the branch. In order to be able to capture different wind directions, a lake might be modelled by several branches in different directions. A method for this, the so called waggon-wheel structure, is proposed in (Alvarez and Verwey, 2000)..
There are some basic suggestions presented about the location of the branches in order to catch the effect of the wind in the best possible way. The branches are located in a so called “waggon wheel” structure: the branches meet at a central point (of the wheel). It is suggested to use at least 6 branches that enclose at most 90 degrees.

Case study

The IJsselmeer is an artificial lake created by the construction of the Afsluitdijk. Its surface together with the Ketelmeer is about 1200km^2 and its average water depth is 2 m. A map of the region is shown in Fig. 1.

Figure 1. Map of the IJsselmeer region

Methods

Determining the branch locations

The following guidelines are summarized in (Alvarez and Verwey, 2000).
1. Nodes should be placed in the deepest areas
2. Horizontal currents can be represented by triangles
3. Internal nodes 6 incoming branches
4. Water should be allowed to flow freely in all directions
5. The total storage area connected to the branches should be equal with that of the lake

The location of the branches will be determined based on these guidelines. Different constructions are going to be made based on the guidelines above, starting from the most simple approach until including a more complex branch network. Some other factors can also be included like making the braches coincide with the main shipping routes. The effect of the wind will be examined by comparing the simulated water levels to measurements.
Fig. 2 shows the bottom level of the IJsselmeer including a configuration of the proposed branches.

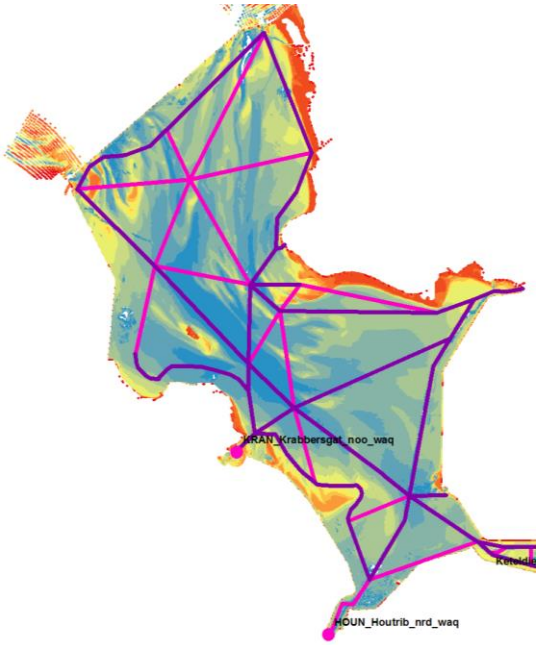

Figure 2. Bottom level of the IJsselmeer and the proposed branches: with purple the shipways and with pink the other branches

Creating the profiles for the 1D model

The profiles are created with the software WAQ2prof (Deltares, 2012). This program creates simplified cross sectional profiles based on the DTM and the results of a two dimensional flow calculation. Each cross section represents the roughness and the volume of water of the surrounding area. The benefit of this approach is that the cross sections are reproducible.

In order to create these cross sections, areas belonging to each cross section should be defined. An example for these areas is shown in Fig 3. It can be seen that the river branches have more cross sections, while the lake part is proposed to be modelled with branches enclosing an angle with each other.

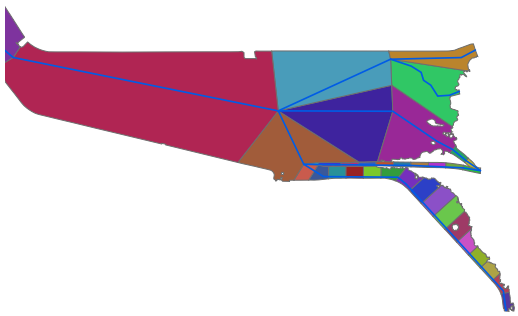

Figure 3. The Ketelmeer, with the proposed model branches (blue) and the areas used to define cross sections

The WAQ2Prof software is mainly developed for river applications and will therefore be improved to be able to generate good profiles for water bodies with considerable storage areas (like lakes) as well.

Representation of the wind in the 1D model

The system is modelled with the one-dimensional hydrodynamic software package Sobek 3 (Deltares, 2013).

The wind speed and direction is spatially uniform, but can be varying in time, while the wind shielding in Sobek 3 is spatially varying, but uniform in time. The only possible inputs are the wind velocity (m/s), the wind direction and the wind shielding. The other coefficients are given as default.

$$\tau_{wind} = -\rho_{air} C_{wind} u_{wind}^2 \cos\left(\varphi_{wind} - \varphi_{channel}\right)$$

where

$$C_{wind} = \alpha_{wind,1} + \alpha_{wind,1} u_{wind}$$

where

τ_{wind}	the wind shear stress (N/m^2)
ρ_{air}	air density (1.205 kg/m^3)
C_{wind}	wind friction coefficient with the coefficients α_{wind1}=0.0005 m/s α_{wind2}=0.00006 s/m
u_{wind}	wind velocity
$\rho_{channel}$	clockwise angle between the positive x-axis of the 1D channel and the North
ρ_{wind}	clockwise angle between the wind direction and the North

At first, the wind hiding factor is kept constant, in order to see the effect of the other factors. Using existing wind measurements, the only other factor determining the effect of the wind is the angle it encloses with the branches, with other words the location of the braches determine how the effect of the wind will be captured.

Results

The building of the model is an on-going process. First, the most simple arrangement of the branches has been proposed. An example cross section calculated from the proposed branches is shown in Fig. 4.

In the continuation of the work cross sections should be calculated for the whole system, and a whole one-dimensional model should be built. After building the model it is validated using measurements of four different storm periods.

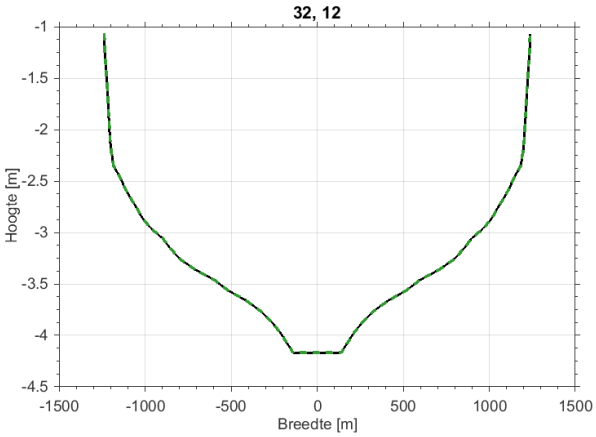

Figure 4. Cross section for one of the branches of the Ketelmeer

Conclusion and future work

One dimensional models of river systems can also include two dimensional water bodies. In case of shallow lakes, which are frequently present in the Netherlands, the effect of the wind on water levels is crucial. It is therefore beneficial to have a methodology developed and tested in order to be able to include these effects to one-dimensional models.
The effects of the schematization are assessed by comparing the results of a basic and a more elaborated branch network. Future work includes sensitivity analysis for wind hiding factor and the wind measurement location.

The developed approach will be later on also applied to other lakes like the Markermeer, Volkerak-Zoommeer and Veluwerandmeren.

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Emergence of different river dynamics through changing vegetation patterns

M. van Oorschot[1,2], M.G. Kleinhans[2], G. Geerling[1], H. Middelkoop[2], T. Buijse[1], E. Mosselman[3,4]

[1]*Department of water quality & ecology, Deltares, PO Box 177, 2600 MH, Delft, the Netherlands*

[2]*Faculty of Geosciences, Universiteit Utrecht, PO Box 80115, 3508 TC, Utrecht, the Netherlands*

[3]*Department of river dynamics and inland water transport, Deltares, PO Box 177, 2600 MH, Delft, the Netherlands*

[4]*Faculty of Civil Engineering and Geosciences, Delft University of Technology, PO Box 5048, 2600 GA, Delft, the Netherlands*

Introduction

Riparian vegetation interacts with morphodynamic processes in rivers to create distinct habitat mosaics supporting a large biodiversity. The pattern of vegetation on the floodplain is determined by hydro-morphological tolerances of the vegetation which in turn are determined by species specific traits (Gurnell, 2013).

Until now, modelling of interactions between vegetation and morphodynamics has been done with a simplistic approach to vegetation without growth and mortality and does not contain dynamic vegetation properties (Bertoldi et al., 2014; Crosato and Saleh, 2011; Nicholas, 2013).

However, to predict accurate floodplain evolution on the long-term, dynamic vegetation processes must be coupled to advanced morphodynamics (Camporeale et al., 2013; Curran and Hession, 2013).

The objective of the study was to use a model to investigate the effect of dynamic vegetation with a range of different strategies and traits on the morphodynamics and river pattern of a meandering river.

Method

We constructed a dynamic vegetation model coupled to a morphodynamic model (Delft3D). The vegetation model includes three classes of vegetation processes: colonization, growth and mortality (Figure 1). A multi-species approach is used with two vegetation types loosely based on *Salicaceae* species.

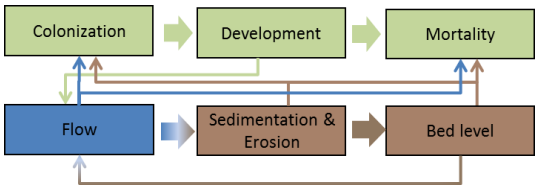

Figure 1. Flow diagram of model processes and interactions.

Colonization takes place depending on the timing of seed dispersal and the water levels during that period. The location for colonization is on bare substrate between the highest and lowest water levels during the annual dispersal period.

Growth of vegetation is calculated based on initial shoot size and diameter and a growth increment per year. When the vegetation survives, its age increases each subsequent year until the maximum age is reached. Depending on the life stage which is related to age, the characteristics of the vegetation are different. Vegetation interacts with the flow through hydraulic resistance calculated by the Baptist equation (Baptist et al., 2007) using the frontal area and the shoot height of the vegetation (Equation 1).

$$C = \frac{1}{\sqrt{\frac{1}{C_b^2} + \frac{C_D n h_v}{2g}}} + \frac{\sqrt{g}}{\kappa} \ln\left(\frac{h}{h_v}\right) \qquad (1)$$

Where C is the Chezy value of the vegetation ($m^{1/2}/s$), C_b is the Chezy value for the un-vegetated parts, C_D is the drag coefficient, n is the vegetation density (stem diameter x number of stems $/m^2$), h_v is the height of the vegetation (m), h is the water depth (m), κ is the Karman constant (0.41) and g is the gravitational force (9.81 m/s).

Mortality of vegetation depends on days of subsequent flooding, days of subsequent desiccation, uprooting, scour and burial.

Several dynamic vegetation scenarios were tested with a range of different species traits and strategies that influenced vegetation settlement location and survival. The scenarios contained species with: 1) fast growth and early death, 2) slow growth and late death, 3) sensitive seedlings, 4) resistant seedlings, 5) low drag, 6) high drag, 7) early dispersal, 8) late dispersal, 9) high vegetation density and 10) low vegetation density.

Results

The results show that different vegetation trait sets and strategies generate a range of different river morphologies and vegetation covers (Figure 2).

Vegetation and morphodynamic statistics of these scenarios reveal a quantitative interaction between vegetation cover, vegetation dynamics and morphodynamics. Increasing vegetation cover and vegetation dynamics decreases sediment transport rates. Furthermore, when vegetation is split up in age classes, the bush stage of the vegetation has the strongest negative effect on the sediment transport rate.

Discussion

Vegetation types containing different morphological features, settling behavior and survival create distinct patterns in vegetation and resulting river morphology. This shows that species traits and strategies are important drivers in creating vegetation patterns and subsequently shaping river morphology. This is also indicated by (Kui et al., 2014) in a large flume experiment with two different vegetation types showing differences in their sensitivity for morphodynamic pressures and resulting survival.

We find a quantitative relation between vegetation cover, vegetation dynamics and sediment transport rate. This is caused by increased hydraulic resistance leading to decreased flow velocity and focusing of flow into the main channel. The area over which sediment transport can take place decreases, leading to a total decrease in sediment transport.

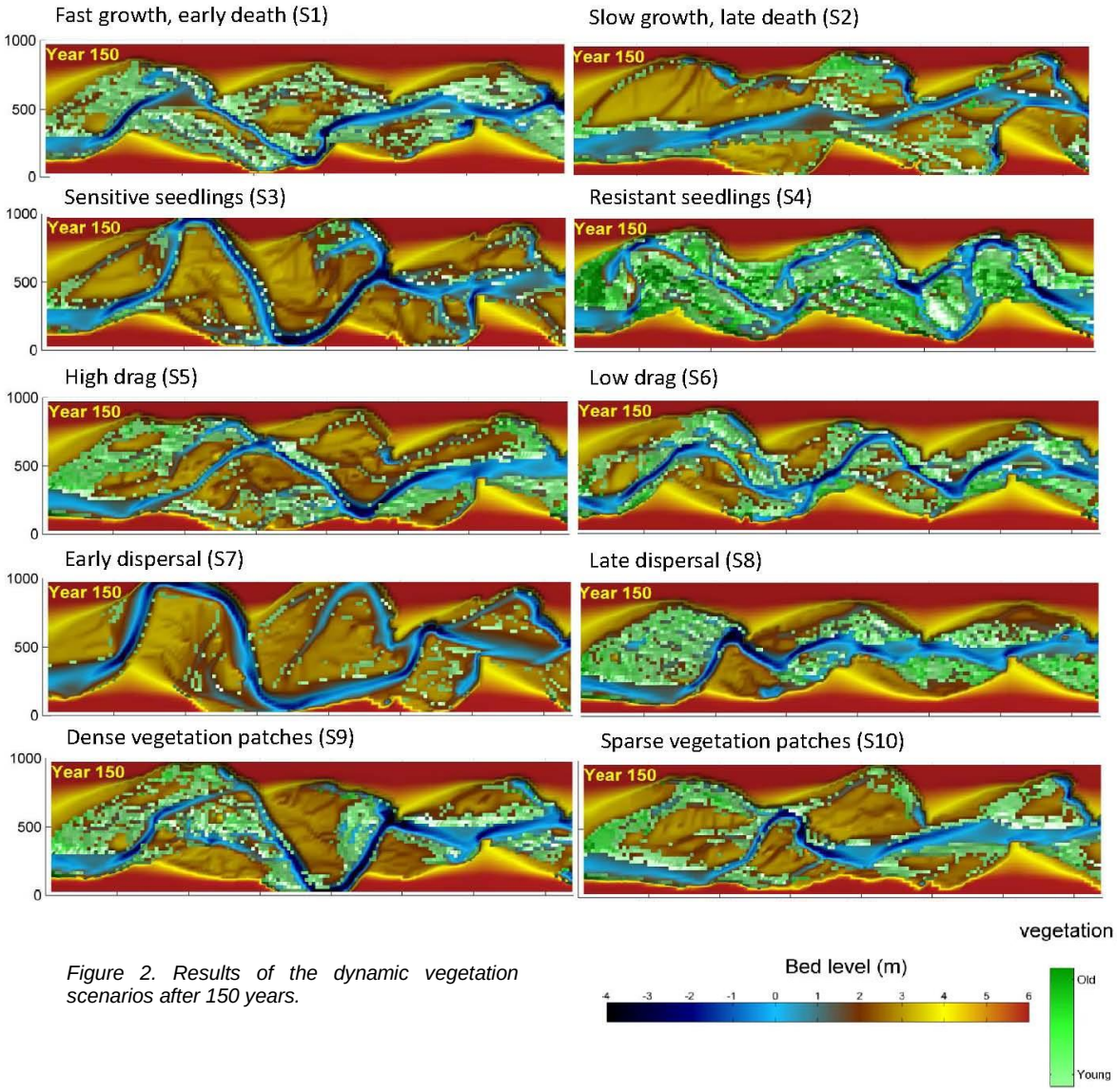

Figure 2. Results of the dynamic vegetation scenarios after 150 years.

We find that the bush-stage of the vegetation has the largest effect on the morphodynamics, which is a reflection of the largest frontal area and therefore the largest hydraulic resistance. This shows that different vegetation stages containing different morphological properties and dynamics have a different influence on morphodynamics and are therefore important to take into account when modelling the interaction between vegetation and morphodynamics. Our model is one of the first models that incorporates such processes and to cater the need of the research community concerning the realistic description of riparian ecosystems (Bertoldi et al., 2014; Camporeale et al., 2013; Kui et al., 2014).

Acknowledgements

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The use of species sensitivity distributions and monitoring to predict the ecological effect of longitudinal training dams

F.P.L. Collas[1], A.D. Buijse[2], A.J. Hendriks[1], R.S.E.W. Leuven[1]
[1]*Radboud University Nijmegen, Institute for Water and Wetland Research, Department of Environmental Science, P.O. Box 9010, 6500 GL Nijmegen, The Netherlands*
[2]*DELTARES, Department of Freshwater Ecology and Water Quality, P.O. Box 177, 2600 MH Delft, The Netherlands*

Introduction

Currently, longitudinal training dams (LTDs) are constructed in the river Waal near the city Tiel (The Netherlands). Traditional groynes are removed and replaced by LTDs which lie parallel to the river edge (Fig. 1). The LTDs are expected to positively influence navigation,

Figure 1: Cross section of a river with on the left bank traditional groynes and on the right bank a longitudinal training dam.

maintenance costs and safe discharge of water and ice. Due to the parallel nature of LTDs, they are also expected to create refuges from dynamics caused by water displacements and waves of passing ships, possibly exerting a positive influence on species assemblages. Moreover, several parameters (e.g., oxygen, temperature) may become more favourable for riverine species due to decreased dynamics in flow velocity.

In order to fully understand, and even predict, the effect of these changing river conditions on species assemblages, species sensitivity distributions (SSDs) are being constructed. SSDs describe the variation among species in their sensitivity to an environmental factor. Using SSDs spatial and temporal predictions of the biodiversity can be made if data is available on 1) the limitation of species' resilience, and 2) the actual level of limiting environmental factors. Here, we present a spatio-temporal effect prediction of temperature, desiccation and salinity on native and non-native mollusc assemblages in the littoral zone of the river Rhine.

Extremely low discharge events and high water temperature events in north-western European rivers are expected to become more frequent and intense due to climate change (Van Vliet et al., 2013). In addition, the salinity of river water may increase due to lower discharges and stronger tidal influences in estuarine areas caused by sea level rising (Verbrugge et al. 2012). These changing conditions affect riverine biodiversity. This can be either a shift from cold-water to thermophilic species or, depending on the conditions, mortality of aquatic species due to desiccation induced by air exposure (Collas et al., 2014).

The mollusc assemblages of the river Rhine are dominated by non-native mollusc species. These non-native species have had a profound impact on biodiversity and ecosystem functioning. Understanding the responses of native and non-native species to changing river conditions will enable model predictions of climate change impact on riverine biodiversity and ecosystems.

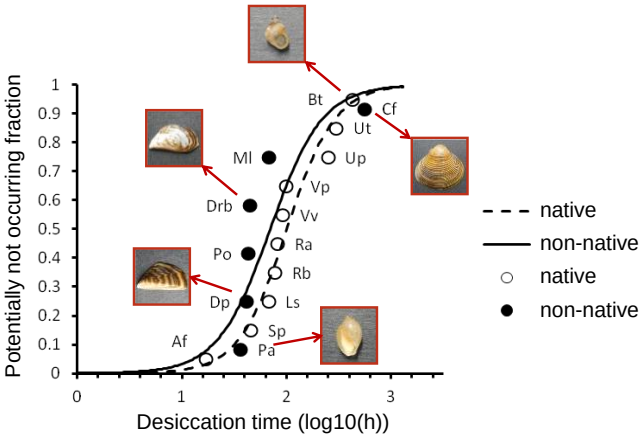

Figure 2. SSD based on the desiccation sensitivity of mollusc species, Abbreviations: Af: Ancylus fluviatilis; Pa: Physella acuta; Dp: Dreissena polymorpha; Drb: Dreissena rostriformis bugensis; Sp: Stagnicola palustris; Ls: Lymnaea stagnalis; Ml: Mytilopsis leucophaeata; Rb: Radix balthica; Ra: Radix auricularia; Vv: Viviparus viviparus; Vp: Valvata piscinalis; Up: Unio pictorum; Ut: Unio tumidus; Bt: Bithynia tentaculata, and Cf: Corbicula fluminea. (adapted from Collas et al., 2014).

Method

Mollusc SSDs were derived for desiccation, temperature and salinity (Collas et al. 2014; Verbrugge et al. 2012; Fig. 2). The mean

and standard deviation of the derived SSDs were used to analyse the spatiotemporal trends of desiccation, water temperature, salinity and their combined effect on mollusc species in the littoral zone of the river Rhine (i.e., a groyne field near Lobith).

These effects were expressed as the potentially not occurring fraction (PNOF) of species. The combined effects of desiccation, temperature and salinity were calculated according to eq. 1:

$$(1) \qquad PNOF_{DTS} = 1 - (1 - PNOF_D) * (1 - PNOF_T) * (1 - PNOF_S)$$

where *D*, *T* and *S* indicate desiccation, temperature and salinity, respectively. Subsequently, a model was constructed that depicts the spatiotemporal variation of combined effects of these stressors on potential species occurrence in the littoral zone. The water level, temperature and salinity data used in the model were obtained from a web-based portal (www.waterbase.nl; Fig. 3).

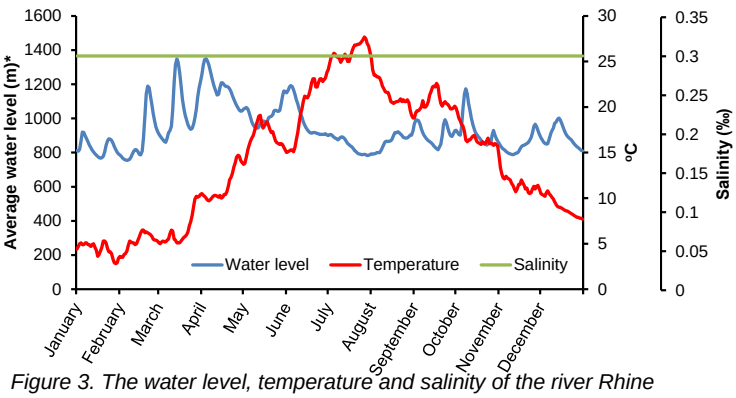

Figure 3. The water level, temperature and salinity of the river Rhine at Lobith during the year 2006 (Above average sea level).*

Results and discussion

Comparisons of the model predictions with field data on species occurrence during the period 1988-2003 revealed that the combined PNOF explained 62 and 80% of the actual not occurring fraction for native and non-native species, respectively (data not shown).

Predictions for a year with extremely low river discharges show that the combined effect of desiccation, temperature and salinity frequently limits the mollusc species occurrence in the littoral zone of the river Rhine at Lobith. However, this effect is due to desiccation and temperature limitation since there was no salinity based limitation at this location. The combined effect is higher for native species than for non-native species.

Native molluscs were additionally limited during June, July and August (Fig. 4A) opposed to non-native species (Fig. 4B). This difference is caused by the lower water temperature tolerance of native molluscs compared to non-native species (Verbrugge et al., 2012). A comparison of the combined PNOF results (Fig. 4) with the water level throughout 2006 (Fig. 3) reveals that the combined PNOF follows the same pattern as the water level. This similar pattern between the combined PNOF and water level indicates that desiccation is the primary environmental factor that determines diversity and composition of native and non-native mollusc assemblages.

The next step is to include other environmental factors (e.g. flow velocity, oxygen availability) and other species groups in the SSD-model. Moreover, field

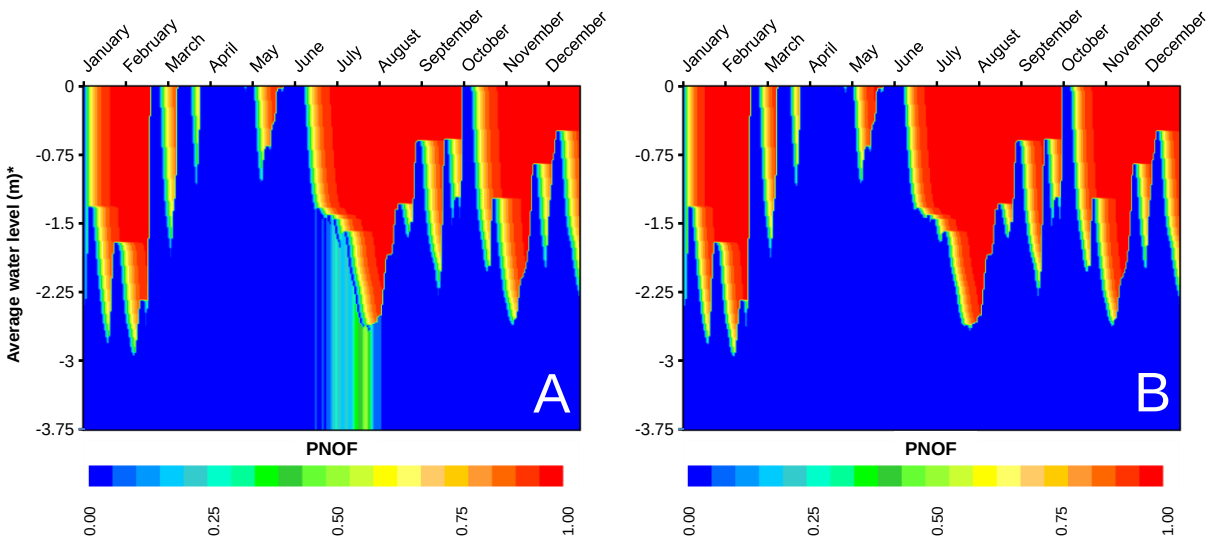

Figure 4. Potentially not occurring fraction (PNOF) due to the combined effect of desiccation, temperature and salinity on (A) native and (B) non-native molluscs in the littoral zone of the river Rhine at Lobith in the year 2006).

measurements of environmental factors and species occurrence will enable additional validation of the spatial and temporal model. Therefore, an extensive field monitoring campaign will be initiated in river sections with LTDs.

The focus of this campaign is twofold: 1) environmental factor measurements near sampling sites of species; 2) measurements of environmental factor levels throughout a year. Species will be sampled at a total of 12 sites of which 6 are located near an LTD. The other 6 sites are situated near other river structures (e.g. side channels, groynes and rip-rap banks; Fig. 5). At each site as many different habitats as possible will be sampled to get a good idea of the total species richness. The focus of the monitoring campaign is based on fish, snails, mussels, crayfish and amphipods. Flow velocity, water temperature, sediment concentration and type, water level, depth, oxygen availability and salinity will be monitored.

Conclusion

- The combined PNOF has a high explanatory value, indicating that SSDs are a useful tool to model species assemblages.
- Desiccation is the primary environmental factor that determines diversity and composition of the mollusc species pool in the littoral zone of the River Rhine.
- Throughout a year, native species are more limited by the three environmental factors than non-native species.
- The SSD approach can be used to predict the effect of climate change for the entire river continuum (from head waters to estuaries) or to determine the relative importance of each environmental factor at different river sections.
- The concept can also be applied for other river systems when river specific tolerance data of species are available.

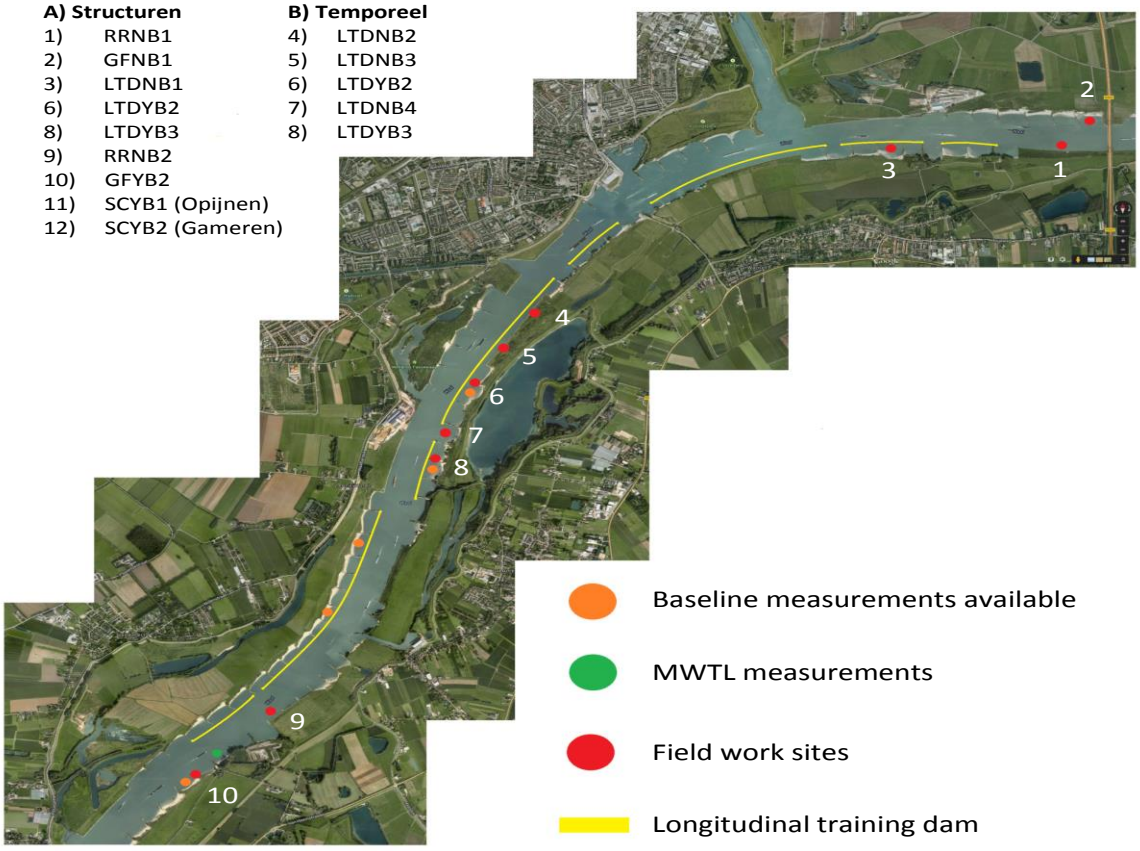

Figure 5. Overview of the different monitoring sites that will be monitored during the monitoring campaign.

Acknowledgements

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The results of this study have also been presented at the 2nd edition of the I.S. Rivers conference in Lyon (France).

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Biodiversity changes in fifteen years of restoration of large multifunctional rivers

M.W. Straatsma[a,c], A.M. Bloecker[b], H.J.R. Lenders[b], R.S.E.W. Leuven[b], M.G. Kleinhans[b]
[a]Faculty of Geosciences, Department of Physical Geography, Utrecht University, PO Box 80115, 3508 TC, Utrecht, The Netherland [b]Faculty of Science, Department of Environmental Science, Radboud University Nijmegen, P.O. Box 9010, 6500 GL Nijmegen, The Netherlands, [c]Corresponding author: m.w.straatsma@uu.nl, tel. +31 (0)30 2532754

Introduction

The last two decades featured numerous river restoration projects with multiple objectives, such as increasing flood safety, biodiversity conservation and improving landscape quality in order to counteract the effects of higher peak discharges and the decline of fluvial biodiversity (Bernhardt et al. 2005). Hydrodynamic models are routinely applied to determine flood safety, unlike biodiversity models, which are only applied in the planning phase of physical reconstruction projects. A long term overview of changes in biodiversity is often not available for lack of input data, and because tools to assess changes in biodiversity over large areas with sufficient detail are missing.

The objective of this study was to determine the changes in biodiversity for protected and endangered species between 1997 and 2012 due to land cover changes and changes in species presence. The study was carried out in the distributaries of the Rhine River in the Netherlands

Methods

To reach the objective, we implemented the BIOSAFE biodiversity model (Lenders et al. 2001; De Nooij et al. 2004) in the Python programming language and extended it, enabling spatiotemporal application to large areas. Four ecotope maps (1997, 2005, 2008, 2012) provided the changes in land cover, and the National Database Flora and Fauna (NDFF) included three million field observations of species in seven taxonomic groups between 1993 and 2014. BIOSAFE was subsequently run for 179 floodplain sections covering the three distributaries of the River Rhine.

Results

The actual biodiversity index, which represented species presence, increased by 100%.The median potential biodiversity index increased by 15%. Of all floodplains, 82% showed an increase in biodiversity. Large variations were found between taxonomic groups and between floodplains. We show that biodiversity of birds, mammals and herpetofauna (amphibians and reptiles) significantly increased in the 5% of the river sections where natural vegetation succession was allowed in more than 15% of the floodplain area. The 5% of the river sections where floodplains were rejuvenated (setting back vegetation succession, and creation of side channels) over more than 5% of the area shows a significant increase in fish and dragon- and damselflies.

Discussion and conclusions

These results show, for the first time and at a large spatial scale, that the biodiversity of protected and endangered species increases due to the floodplain interventions carried out with multiple objectives. Additionally, they suggest that a range of different measures at different times is required for the highest biodiversity across taxonomic groups (i.e. creation of natural riverine landscape heterogeneity). Biodiversity changes represented conservative estimates as classification errors in the ecotope map, and positional inaccuracies of the field observations of species partly obscured the species dynamics.

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Suitability of landscape classification systems for quantification of ecosystem services in BIO-SAFE

K.R. Koopman[*1], D.C.M. Augustijn[2], A.M. Breure[1,3], H.J.R. Lenders[1], R.S.E.W. Leuven[1]
[1] Radboud University, Department of Environmental Science, Institute of Water and Wetland Research, P.O. Box 9010, 6500 GL, Nijmegen, The Netherlands
[2] University of Twente, Twente Water Centre, P.O. Box 217, 7500 AE, Enschede, The Netherlands
[3] National Institute for Public Health and the Environment (RIVM), Centre for Sustainability, Environment and Health, P.O. Box 1, 3720 BA, Bilthoven, The Netherlands
** Corresponding author; e-mail: k.koopman@science.ru.nl*

Introduction

In the last century, rivers have been heavily regulated to maintain and improve important river functions such as navigation, water supply and biodiversity. These functions are increasingly threatened by climate change and population growth, asking for shifts in river management strategies. The multidisciplinary RiverCare programme was initiated to develop tools and measures for creation of more self-sustaining rivers and reduction of management costs (RiverCare, 2013; Hulscher et al., 2014). Management cost reduction requires tools for evaluation of the effectiveness of riverine management measures.

RiverCare subproject E2 focusses on ecosystem services of river-floodplain systems. The goal is to develop an assessment tool for evaluation of effects of river management measures on the spatiotemporal development of riverine ecosystems and their services (Fig. 1.). Starting point for development of such a tool is BIO-SAFE, a model by which the effects of river management on biodiversity can be assessed (Lenders et al., 2001; De Nooij et al., 2004).

BIO-SAFE is an excel-based model that predicts and valuates biodiversity of a study area. It links landscape ecological units (ecotopes) to (potential) presence of certain target species and can be used for calculation of actual and potential biodiversity values of river-floodplain systems at various spatial scales. The biodiversity values are calculated using criteria that relate to the political and legal conservation status of target species. Management induced changes in surface areas and types of landscape units result in changes of biodiversity values, allowing the evaluation of management measures by comparing before and after values (Lenders et al. 2001; De Nooij et al., 2004).

Incorporating ecosystem services into BIO-SAFE requires linkage of these services to standardized landscape units from a landscape classification system. An extensive literature review was performed to determine the suitability of existing landscape classification systems applicable for linkage to ecosystem services of river systems (green block in Fig. 1.).

The aim of this research was I) to review landscape classification systems that are used across the globe; and II) to determine which landscape classification systems are applicable to rivers and most suited for linking and quantifying spatiotemporal developments of riverine ecosystem services in relation to management measures.

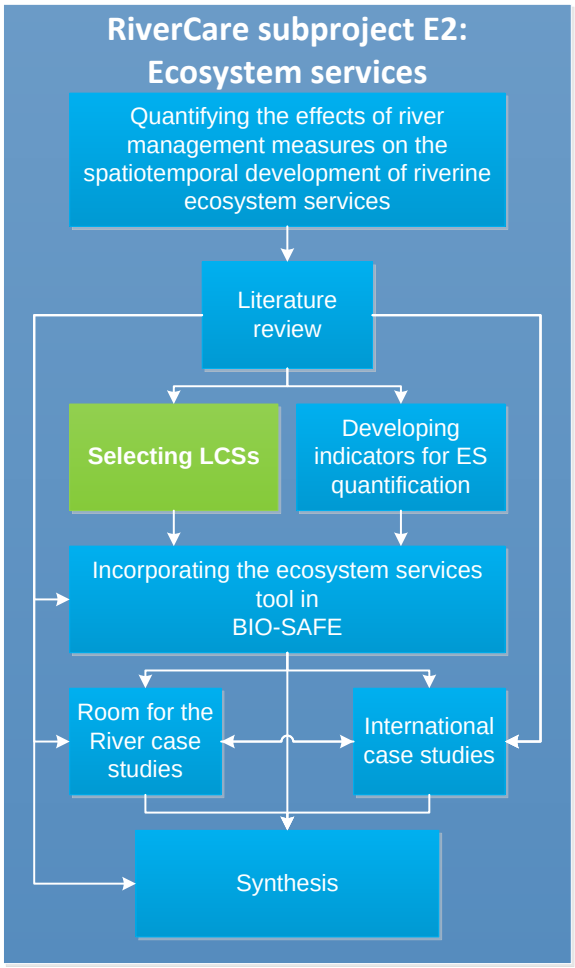

Figure 1. The different steps of RiverCare subproject E2. The place of this review in the project is indicated by the green block: Selecting suitable landscape classification systems (LCSs). (ES = Ecosystem services).

Methods

A search for peer reviewed literature on landscape classification systems and their links to ecosystem services was performed using ISI Web of Knowledge (www.isiknowledge.com) and a set of search terms related to landscape classification, river systems and ecosystem services. All hits were screened for further selection. The landscape classification systems had to comply with our definition: 'a landscape classification system describes the landscape in multiple classes or features that are distinctive from each other and spatially explicit'. Suitable papers were reviewed and analysed using criteria such as potential for linkage to ecosystem services, applicability to riverine landscapes and suitability for studying spatiotemporal development of landscapes and their ecosystem services.

Results

Out of the 546 hits, we selected 95 papers (that often included multiple case studies) for further analysis. In total, 31 (33%) papers linked ecosystem services to landscape units using either quantitative, semi-quantitative or both types of methods (Fig. 2). Quantitative methods expressed ecosystem services in biophysical or monetary units, while semi-quantitative methods gave scores indicating the potential of landscape units to deliver ecosystem services. The first paper that linked ecosystem services to various landscape units was published in 2002 (Konarska et al., 2002). It took several years before approaches to link ecosystem services to landscapes emerged in other papers. In 2005 a slight increase in publications linking ecosystem services to landscape classifications was visible, followed by a rapid increase after the years 2010 / 2011. Only 25 (26%) of the selected papers applied landscape classifications to rivers.

We found several landscape classification systems that are suitable for incorporation in the new BIO-SAFE ecosystem services tool. Table 1 provides three examples that allow classification on different scales: 1) CORINE covers most of Europe (EEA, 1995), and is applicable to rivers. Several times it has been linked to ecosystem services and it is already incorporated in BIO-SAFE. 2) The Dutch RWES classification (Rijkswateren-Ecotopenstelsel) was specifically designed to classify Dutch river systems (Van der Molen et al., 2000; Willems et al., 2007) and is also incorporated in BIO-SAFE. So far, the RWES classification has not been linked to ecosystem services. 3) The GLC2000 has a global coverage and can also be applied to rivers (Mayaux et al., 2006). It can be incorporated into BIO-SAFE and has been linked to ecosystem services. Furthermore, the GLC2000 was used in the Millennium Ecosystem Assessment of the United Nations (MEA, 2005). Only CORINE has been used to study the effects of landscape changes on ecosystem services performance (Scollozi et al., 2012), but the RWES and GLC2000 classifications are also considered suitable.

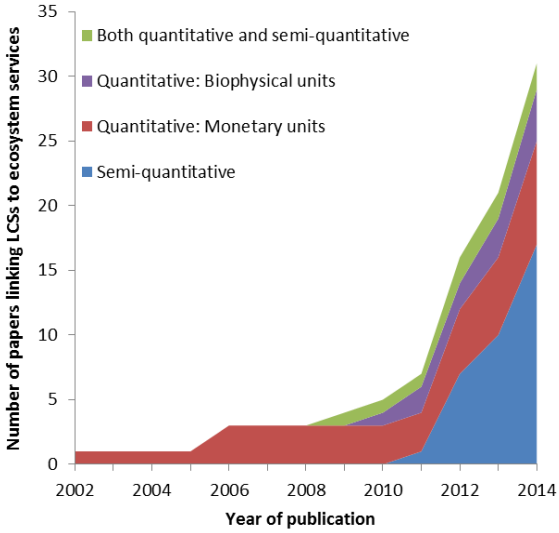

Figure 2. The cumulative number of papers that linked ecosystem services to landscape classification systems (LCSs), using different methods (see legend in figure).

Table 1. Landscape classification systems that can be applied to river systems, incorporated in BIO-SAFE and linked to ecosystem services (1 = Is the LCS applicable to rivers-floodplain systems?; 2 = Is the LCS already incorporated in BIO-SAFE?; 3 = Have ecosystem services been linked to LCS?; 4 = Has the LCS been used for studying effects of landscape changes on ecosystem service performance?) (+ = yes; - = no; (+) = considered suitable for this category).

Name	Rivers[1]	BIO-SAFE[2]	ES[3]	Spatio-temporal ES[4]
CORINE	+	+	+	+
RWES	+	+	- (+)	- (+)
GLC2000	+	- (+)	+	- (+)

Discussion

We reviewed landscape classification systems that are used worldwide for a wide range of purposes and different types of landscapes. About 26% of these landscape classification systems were applied to riverine landscapes. Most of these systems classify both the terrestrial and aquatic part of the river system. However, some only focussed on either the aquatic or terrestrial part of the river system. Only six of these river-applicable landscape classification systems were linked to ecosystem services.

The number of papers linking ecosystem services to landscape

classification systems increased significantly after the publications of major international ecosystem services works such as: the Millennium Ecosystem Assessment in 2005 (MEA, 2005), the TEEB reports in 2010 (TEEB, 2010) and the CICES classification in 2011 (Haines-Young and Potschin, 2011) (Fig. 2). Initial linking was only quantitative using monetary units. The publication of these major works provided new approaches and methods that were picked up by the scientific community. Especially the semi-quantitative approach was used often in recent studies (Fig. 2). In this approach, expert judgement was used to determine the landscapes' capacity to deliver ecosystem services. The use of this method offers a relatively 'quick and easy' approach to identify the ecosystem services of a given study area. Furthermore, it allows for comparing the delivery of different ecosystem services among each other. However, the method also possesses a level of subjectivity which, on the other hand, might also be useful when the experts involved are also stakeholders in the area. It thus enables the incorporation of different stakeholders' views in the ecosystem services assessment. The number of papers describing biophysical quantification of ecosystem services through indicators was very limited.

We found several landscape classification systems that are suitable for the development of an ecosystem services tool in BIO-SAFE, e.g. the systems listed in Table 1. More suitable classification systems exist and are described extensively in Koopman et al. (in prep.) The main advantage of using CORINE and the RWES classification is that they are incorporated already in BIO-SAFE. Both systems are applicable to rivers. CORINE has already proven to be suitable for biophysical quantification of ecosystem services (Burkhard et al., 2009). Furthermore, CORINE is applicable across Europe to multiple scales (continental, national and regional) allowing the ecosystem services tool to be applied at an international level. However, since CORINE's resolution might be rather coarse for application on regional scales, it is an option to use the RWES classification for this level of scale. Preliminary results indicate that it is feasible to link the ecological landscape units of the RWES classification to indicators for various ecological functions and the state of ecosystem services. Since, the RWES classification was designed to classify Dutch river-floodplain systems, direct application in other countries might prove difficult due to, for instance, missing specific landscape types (e.g. braided rivers or mountainous headwaters). However, it should be achievable to extend the RWES classification by incorporating additional ecotope types. The GLC2000 might be useful for assessment of ecosystem services at larger scales or in other biogeographical regions than Europe. This landscape classification system is suitable for ecosystem services linking and is also applicable to rivers. It has not been incorporated in BIO-SAFE yet, but considering its landscape classes, there are ample possibilities to do this. For application on national levels, however, the resolution of the GLC2000 might be too coarse (Schulp and Alkemade, 2011). So far, only CORINE has been used to determine effects of landscape changes on ecosystem services (Scollozi et al., 2012). However, it is expected that all three classification systems can take spatiotemporal development of ecosystem services into account using transition matrices.

Although we found several landscape classification systems that are potentially suitable for the development of an ecosystem services tool in BIO-SAFE, knowledge on indicators for biophysical quantification is still very limited and scattered across the literature. Furthermore, we did not find any approaches or case studies that assessed spatiotemporal development of ecosystem services in relation to river management measures.

Future work

Future work will focus on development of biophysical indicators for quantification of ecosystem services and studying how these indicators may be coupled to ecosystem quality and develop spatiotemporally in relation to various types of river management measures (Fig. 1).

Acknowledgements

These results of our study were also presented on the 2nd International I.S. Rivers Conference in Lyon, 22-26 June, 2015

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Interdisciplinary research in riverine sciences: a few feedbacks from 20 years of experiences in Eastern France

H. Piégay
University of Lyon, UMR 5600 EVS CNRS, Site ENS, 15 Parvis R. Descartes 69007, Lyon.
Email: herve.piegay@ens-lyon.fr

Abstract

The present key-note lecture is focused on feedbacks on interdisciplinary research in riverine sciences. Such research strategy has been promoted in France by the CNRS (National Centre of Scientific Research) in the early 1980's, notably on large rivers such as the Seine, the Garonne, the Loire and the Rhône. Such research has been continuously supported by this national institution and in the early 2000's, some "Zones Ateliers" (ZA; e.g. research area) have been created to promote long term interdisciplinary research on the environment (see. http://www.za-inee.org/) related to ILTER (International Long Term Ecological Research) sites network. More recently another interdisciplinary framework has been also created, so called the OHMs – Observatories human / environment (http://www.cnrs.fr/inee/outils/ohm.htm) whose philosophical and strategic research basements are a bit different, more focused on anthropogenic systems and their evolution, but still based on interdisciplinary research effort.

Interdisciplinary research is then institutionalised since 35 years now and 4 generations of researchers are actually involved in these research infrastructures. Since the emergence of the ZA, interdisciplinary research progressively evolved to integrated sciences with a strong interaction with practitioners, opening a new era within which projects and questions are more and more co-constructed by the two communities. The conference IsRivers (http://www.graie.org /ISRivers/) which occurs every three years in Lyon (two first editions in 2012 and 2015) has been created to try to promote such approaches at a European level for implementing the WFD.

This evolution fully occurs on the Rhône basin with an emerging research team in the 1980s who created a ZA (the ZABR - http://www.graie.org/zabr/index.htm) in 2001 (Fig. 1) and an OHM for the Rhône corridor (http://ohm-vr.org/) in 2010.

Key-questions progressively evolved over the period. In the 1980s the group mainly focused on diagnosis and monitoring the effects of human impacts, such as damming. They promoted the emerging concept of hydrosystem and the multi-scalar view of rivers (Amoros and Petts, the fluvial hydrosystem, 1993). Following this first period, the group then moved in the 1990s to the restoration issue asking questions such as "What do restoration or rehabilitation mean? Can we restore? How?", clarifying the terms (e.g., historical reference, rehabilitation versus restoration, trajectory and adjustment; see Henry and Amoros, 1995a and 1995b; Dufour and Piégay, 2009) and improving the practices in term of monitoring, definition of indicators, design of response models and identification of evidences of success, sustainability (forms versus processes, climate change) (see for example the Special Issue published in 2015 in Freshwater Biology about the Restoration of the Rhône or Morandi et al. 2014).

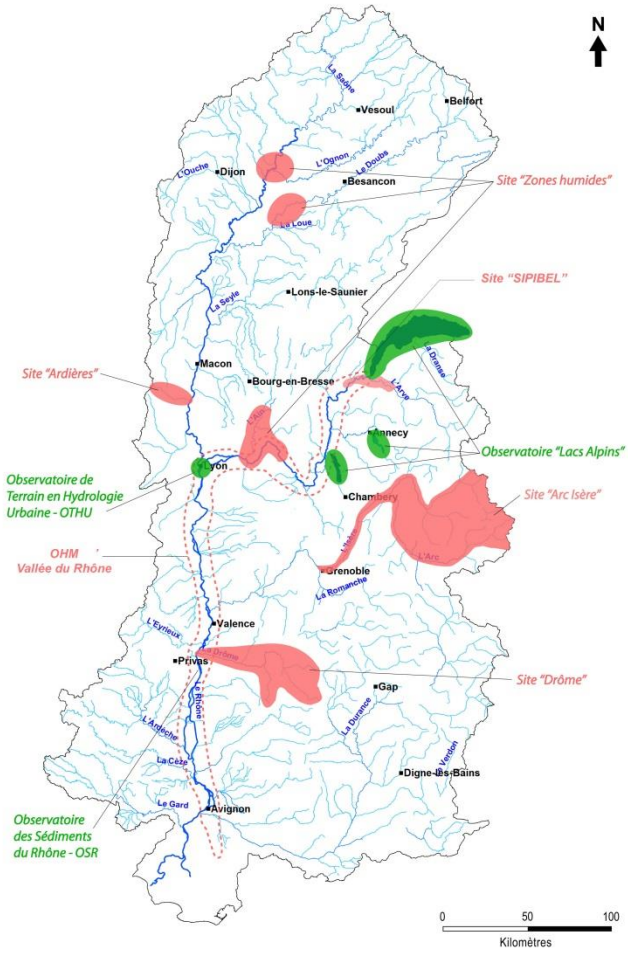

Figure 1. The Rhône basin and the different sites and observatories of the Zone Atelier Bassin du Rhône (the ZABR - http://www.graie.org/zabr/index.htm)

Over the last decade, the group widened its biophysical approaches, evolving social sciences and exploring simultaneously a set of issues such as the historical trajectory of the Rhône, the implementation of a sustainable development policy, the hydrosystem functioning, the environmental risks (pollutants, flooding, climate change and water availability), the restoration policy and new tools development (simulation, participation, visualization, data sharing). The group is now always in debate with stake-holders with whom it builds projects, and it is also progressively more involved with the public. It is working on the emerging concept of integrative sciences applied to riverine systems in relation to complex questions for which a set of knowledge are needed: risks, stakes associated with sediment transfers, public policy assessment (restoration, ecosystem services), design of actions to promote sustainable development.

A few examples to illustrate interdisciplinary research at a project level over a few years or at a larger and long term institutional level will be introduced and discussed. The researcher involved in such projects must be humble and open-minded, ready to overpass language differences to progressively establish a common base with other researchers. He must be convinced that understanding parts of complex environmental systems or questions needs a range of knowledge that a single discipline cannot provide. The group must identify and then share common scientific questions and establish conceptual models to clarify how they see the problem and to share common hypotheses to validate (Fig. 2; see also Pont et al. 2009).

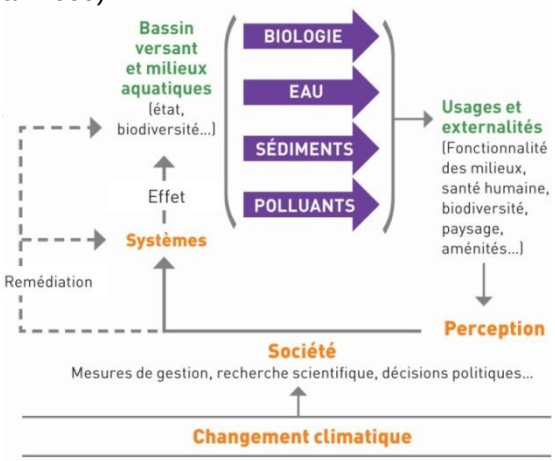

Figure 2. Example of conceptual model on which integrative sciences within the Zone Atelier Bassin du Rhône is funded (http://www.graie.org/zabr/index.htm).

Interdisciplinary can be powerful and useful, and fairly easy to handle when clear/well established scientific questions are posed so that initial brain stormings are then crucial for the success of the next steps. Interdisciplinary is also easy if funding flux is attractive to promote such approach and then the interactions between scientists and practitioners is critical in this context. How practitioners perceive sciences and how they consider scientists as contributors to solve problems is also very important to create a base for applied interdisciplinary and integrative approaches. In best conditions, long term cooperations can be expected allowing to rapidly incorporate recent knowledge in the river management process but it needs a specific regional or national integrative context and funding strategy. Interdisciplinary is also easier if there is a culture of interdisciplinary in the research institutions and effective benefits in careers to go there, but it is not always the case, it depends on national research policies and disciplines. With the development of integrative sciences, another key question is now emerging on how to produce scientific knowledge when scientists are fully involved in the system they are studying. The scientist is now also an actor as fishermen or unhabitants influencing social debates and solutions chosen. In this domain, interdisciplinary is very important because contrasted view points or perspectives help to reconsider ethics, objectivity and main objectives in scientific practices to answer social needs.

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Implementation and verification of morphology in SOBEK 3

K.D. Berends[1,2], A. L. de Jongste[1,3], M. van der Mheen[1], W. Ottevanger* [1]
[1] *Deltares, Department of River Dynamics and Inland Water Transport, Postbus 177 2600 MH Delft*
[2] *University of Twente, Department of Water Engineering and Management, Faculty of Engineering Technology, P.O. Box 217, 7500 AE, Enschede, the Netherlands*
[3] *Witteveen+Bos, Business Line Deltas, Coasts and Rivers, Group Hydrodynamics and Morphology, P.O. box 2397, 3000 CJ, Rotterdam, the Netherlands.*
** Corresponding author; e-mail: willem.ottevanger@deltares.nl*

Introduction

Rivers form an important part of the landscape. They are important for navigation and the transport of goods. Moreover, the fertile lands, drove people to settle close to the river.
Numerical models play an important role for predicting the hydrodynamic (water levels, velocities, discharge distribution) and expected morphodynamic (bed levels) changes in rivers caused by their natural development or updated measures (e.g. Room for the Rivers for safety from flooding).
Although 3D and quasi-3D models are widely used for morphodynamic studies, for understanding large scale systems 1D models have a clear advantage as they are faster, are less data intensive, and give a good general representation of the river system. Moreover, when input data is uncertain, 1D models are better suited for multiple scenario runs as they are faster.

The ability to calculate sediment transport and morphological change in a 1D river model had been present since the early 90's in SOBEK RE. With SOBEK RE becoming rapidly outdated, early attempts at implementing morphology in SOBEK 2 were compounded by various difficulties.
During the development of SOBEK 3 – based on the same computational core of SOBEK 2 – we successfully coupled the numerical concept of SOBEK RE with the sediment transport libraries of Delft3D. Furthermore, we designed the file-based user interface to be identical, except for some 1D specific components, to Delft3D in anticipation of a graphical user interface suitable for both SOBEK 3 and the Delft3D Flexible Mesh, which is the follow-up of Delft3D with unstructured grid support.

Figure 1 Present interface to morphology module from Sobek 3 en Delft3D Flexible Mesh

In this paper we want to formally introduce the first stage implementation and verification of sediment transport and morphology in SOBEK 3. We will focus on the numerical implementation, user interface and verification test cases. Finally we will discuss the outlook for future SOBEK 3 developments.
We very much welcome suggestions for further improvements of SOBEK and extend our desire for cooperation with willing parties for the further development of this landmark 1D modelling suite.

Numerical implementation

As the time scale of hydrodynamic changes is generally faster than that of morphodynamic changes for low Froude numbers in rivers, it is possible to solve the hydrodynamics and morphodynamics in a decoupled manner.
The solution of the hydrodynamics in SOBEK 3 uses a staggered numerical approach, in which the water level is computed at the cross-sections and in the connection nodes, the discharges are computed at the reach segments (cf. Figure 2). For further details consult the SOBEK 3 technical reference manual (2015).
For the morphodynamics, cross-sections are computed at bed level locations. Contrary to the hydrodynamic solver for the water level, it is possible for the bed level to have different values for each branch near the connection node. For this reason near a connection node a control volume for the cross-section is made for each branch.
To compute bed level changes the Exner balance is computed and near connection nodes a so-called "nodal point relation" is defined.

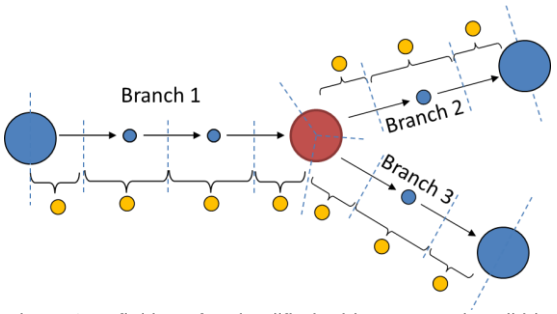

Figure 2 Definition of a simplified grid geometry (small blue dots denote locations of cross-sections, large blue dots are boundary nodes, red dots are connection, nodes and orange dots show the location of the bed level data. The arrows represent the reach segments.

A particular problem in one-dimensional sediment transport is the distribution of sediment at nodes such as bifurcations, trifurcations or more exotic variants. The problem at nodes is that the system has more unknowns than equations when considering only the shallow water equations and the Exner balance. To close the system of equations nodal point relation is introduced.
Presently two possible relationships are implemented: look-up table based or following a functional relation. The look-up table offers the most flexibility but is only possible at bifurcations. It requires the user to specify a table which relates the discharge ratio of the two outflowing branches (Q_2/Q_3) to the sediment transport at those branches (S_2/S_3).
The functional relation is applicable at all nodal permutations. For a bifurcation it is equivalent to the formula proposed by (Wang et al. 1995):

$$\frac{S_2}{S_3} = \left(\frac{Q_2}{Q^3}\right)^k \left(\frac{W_2}{W_3}\right)^m$$

with sediment transport *S*, discharge *Q*, width *W*. Coefficients *k* and *m* are configurable, see next section. If the user defines no explicit relationship, the default relationships shown in Table 1 are chosen.

Table 1 Settings for k and m in the functional nodal relation definition

k-value	m-value	Remark
1	0	'Proportional', default setting
k	*1-k*	Original formulation by Wang et al. (1995)

For those familiar with SOBEK RE we like to add that we have chosen not to implement the linear relationship because it is mathematically asymmetrical and only suitable for bifurcations.

User interface

SOBEK 3 has a modular structure in which one or more modules can be coupled in an integrated model. The SOBEK 3 modules consist of D-Flow1D (channel flow), D-RTC (realtime-control), D-WAQ1D (water quality), D-RainfallRunoff and D-Flow1D2D (2D floodplain flow). Morphology is implemented as an integral part of the 1D channel flow module. Like Delft3D, we have implemented a file based interface. All files have a keyword-based template. We hope to complement the file-based interface with a graphical user interface in Delta Shell (Donchyts & Jagers 2010) as part of future developments.
Any SOBEK 3 model that includes the D-Flow1D module can easily be enabled to include sediment transport and morphological change. Morphology is controlled by two files: the <*.sed> file for sediment characteristics and the <*.mor> file for morphological change. Other attribute files are defined by these two master files following the Delft3D standard.
We deviate from the Delft3D standard on typical one-dimensional particulars. The most important one is the introduction of the 'nodel relations definition file' <*.nrd> in which the modeller defines the distribution of sediment at nodes (bifurcations, trifurcations).
While the Delft3D standard well-known by morphological modellers, the 1D adaptation still has some features that can greatly benefit from feedback of modellers, especially considering spatially variable input.

Besides the file-based interface we have made some aspects of the morphological API available to users via Python scripting. This powerful extension offered by Delta Shell (Donchyts & Jagers 2010) allows for manipulation of the computation, even during simulation. To demonstrate this, we have designed a proof-of-concept implementation of full-fledged dredge- and dump functionality in the open-source Python language. Due to the efficient direct access to the SOBEK computational core, we were able to implement a dredge- and dump function that does not slow down computation significantly, even though it executes each time-step.

Verification

A necessary step of model software development is the verification, which is distinct from operational validation of a river model. For the sake of clarity, we refer to the model cycle of (Rykiel 1996) in figure 1. The aim of verification is to demonstrate that the computer implementation is correct. We want to check both the implementation of physics for

'bugs' and logical errors. To do so, we have designed tests to show SOBEK 3 is capable of certain basic functionality. In this section, we introduce some of these tests, all of which are documented in a dedicated validation document that is available upon request.

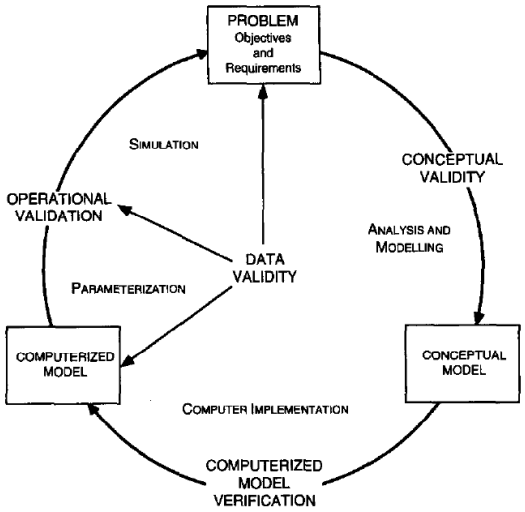

Figure 3 The model process by Rykiel (1996) based on (Sargent 1988) forms the basis of our study

Validation tests have been set up to prove that the simulated sediment transport is correct. The test cases are uniform channels with stationary hydraulic conditions. Validation test have been done for the sediment transport formulas of 1). Engelund and Hansen (1967) and 2). Meyer-Peter and Müller (1948) for various sediment sizes, Chézy roughness and bed slopes. Differences between simulated sediment transport and analytical sediment transport are smaller than $2 \cdot 10^{-5}$ % for Engelund and Hansen and smaller than $2 \cdot 10^{-3}$ % for the Meyer-Peter and Müller.

Another tests have been set up to prove that SOBEK correctly distributes sediment at bifurcations given a certain nodal relation. Validation tests have been done for both the functional relation (Wang, 1995) and the table method. The maximum difference between simulated sediment transport and analytical sediment transport is smaller than $4 \cdot 10^{-5}$ %.

In addition, validation tests have been set up to prove that the simulated morphological changes are correct. The test case consists of propagation of a local shoal on top of an equilibrium bed level in a channel with homogeneous cross-sections morphological acceleration factor. The simulated propagation speed of the bed disturbance have been compared with an analytical approximation which is based on the method of characteristics. The validation tests showed that both the propagation speed of bed disturbances and the equilibrium bed level slope (cf. Figure 4) can be accurately calculated by SOBEK, also for simulations with a higher morphological acceleration factor.

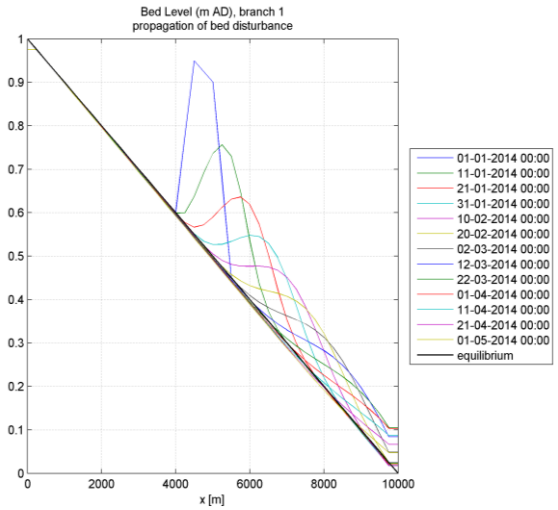

Figure 4 Simulation of the propagation of a bed disturbance.

Conclusion

The morphological extension to SOBEK 3 is fairly new and as of 2015, SOBEK 3 is used in the official curriculum of the University of Twente for educational purposes on hydro- and morphodynamics in rivers, thereby replacing SOBEK RE.

The morphology component in SOBEK 3 now offers full support for total load and bed load transport, in which it is possible to specify spatially varying sediment diameters, which makes it applicable to a wide range of alluvial rivers. Furthermore, some important improvements have been implemented, e.g. access to the well-validated sediment transport library of Delft3D and the better readable file format for the nodal point relations. Furthermore, the scripting toolbox contained within Delta-Shell giving access to the model variables during run-time, offers powerful functionality.

In the future, we hope to extend the morphology component in SOBEK 3 to support to multiple sediment fractions, and also include the effects of suspended sediment. We very much welcome suggestions for further improvements and extend our desire for cooperation with willing parties for the further development of morphodynamics within SOBEK 3.

Acknowledgements

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Exploring dredging strategies on the Parana River (Argentina): morphodynamic modelling for navigation channel maintenance

A.J. Paarlberg [1], F. Huthoff [1], M. Guerrero [2]
[1] *HKV Consultants, Lelystad 8232 AC, The Netherlands, a.paarlberg@hkv.nl, f.huthoff@hkv.nl*
[2] *Department of Civil, Chemical, Environmental and Materials Engineering (DICAM), University of Bologna, Bologna 40136, Italy, massimo.guerrero@unibo.it*

Introduction

Despite the high costs, hindrance to navigation and environmental side-effects, repetitive dredging is commonly employed in rivers to ensure that freight transportation can keep going throughout the year. In this paper we show that hydro-morphodynamic modelling can be used to improve dredge-and-dump activities while mitigating unwanted side-effects deriving from these activities in rivers. As a case study, we focused on dredging activities on the Lower Parana River in Argentina (Figure 1). A more elaborate treatment of this study may be found in Paarlberg et al. (2015).

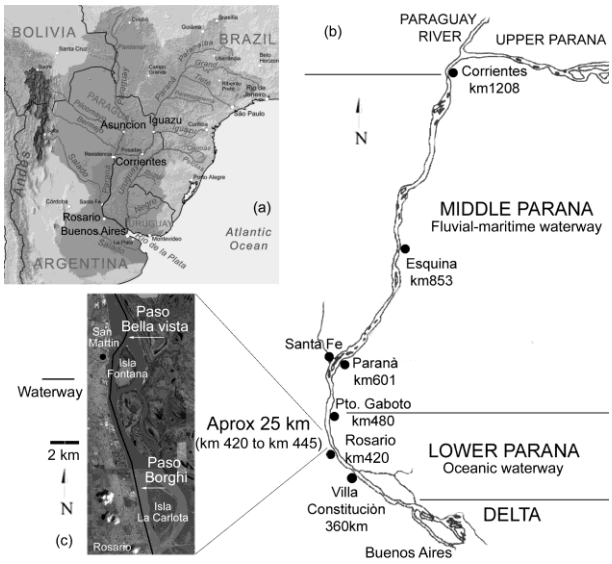

Figure 1. (a) The Parana River section in the "La Plata Basin" watershed, (b) The Parana River and waterway and (c) the study section between San Martin and Rosario (taken from Paarlberg et al, 2015).

Method

The study area is a 25 km river section, approximately between the cities Rosario and San Martin, and consists of a wide main channel and several smaller secondary channels. Figure 1c shows two critically shallow zones in this part of the Parana River: Paso Bella Vista and Paso Borghi. A more detailed map of the bathymetry is shown in Figure 3, also indicating the two shallow zones.

We have set-up a morphodynamic model of this river reach within the open-source modelling Suite Delft3D (Deltares, 2015). The model is largely based on the set-up as described in Guerrero et al. (2015), but includes two key novelties: (i) we incorporated a detailed discharge-hydrograph, as opposed to using yearly-averaged river discharges only and (ii) the dredging activities were explicitly integrated in the computational simulations, allowing continuous interaction between flow and bed response.

Further characteristics of the model are: (i) the shallow water equations are simulated in a horizontal plane (2-DH) (ii) sediment transport is computed by using a single fraction total-load formula of Engelund and Hansen (1967), with 0.26 mm grain size to represent sediment from the river bed and (iii) a morphological acceleration factor (MF) is used to be able to calculate dredging loads over a 6 year period (2004-2009). That is, bed level effects after each hydrodynamic time step are multiplied by a MF to estimate the bed effects on longer (morphological) time scales.

Following e.g. Sloff et al. (2009), a quasi-steady approach was applied to be able to reduce computation time for this six-year time-series using the MF. For this purpose the discharge time-series was discretized into a limited number of discharge levels. In Figure 2 the daily discharge for station Chapeton is shown ("daily Q") for the period 2004–2009. From this daily time-series two discretized, or "stepped", series were derived. First, "yearly Q" gave the yearly-averaged discharge per calendar year. Huthoff et al. (2010) showed the importance of discharge variability in morphodynamic simulations. Therefore, a second discretization, the series "stepped 10-day Q" was derived by smoothing the daily discharge series by taking 10-day averages and then discretizing the smoothed series to five distinct discharge levels (Figure 2).

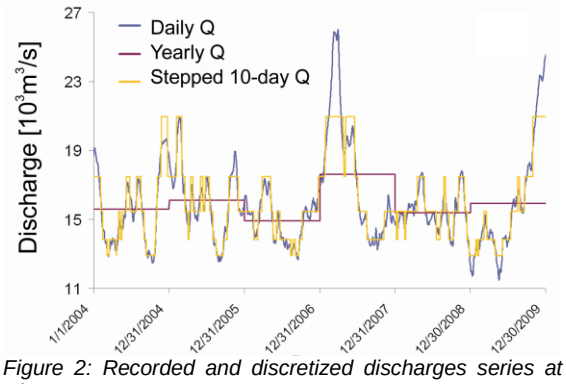

Figure 2: Recorded and discretized discharges series at Chapeton.

The maximum discharge level in the "stepped 10-day Q" was chosen at the 95%-level. At this discharge level the floodplains of the Parana River are just starting to inundate (Q = 20,955 m3/s). Peak values missed were assumed to hardly affect the river channel morphology because of their relative short duration. The lower discharge level was set at the 5% level, where dredging activities can still take place.

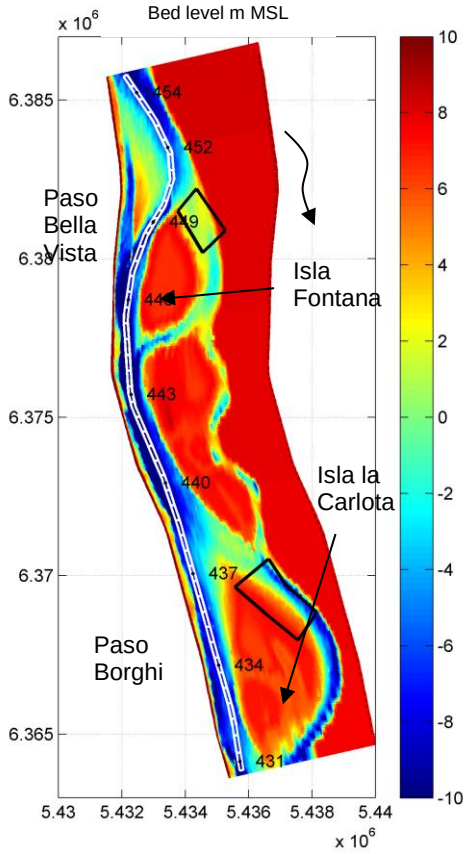

Figure 3: Initial bed level in m above mean sea level (MSL). The white lines indicate the navigation channel, subdivided in 500 m long sections. The black "polygons" indicate dump sites "Bella Vista" and "Borghi".

In the chosen section of the Parana River a navigation channel is maintained that should remain at least 10 m deep at all times (Re et al., 2014; Figure 3). Dunes of mean height and length of 3.5 and 100 m, respectively, have been observed on the riverbed in the study reach (Parsons et al., 2005; Guerrero & Lamberti, 2013). The presence of these dunes can lead to local shallows that need to be dredged as well. Dunes are very dynamic and there is still debate on how to predict their dimensions appropriately, although on the Parana dune heights appear to be significantly correlated with water depth (Guerrero & Lamberti, 2013). Following Paarlberg et al. (2015) we increased the dredging depth with 1,5 m (approx. half the observed dune height) to a dredging depth of 11,5 m in the navigation channel, to take the influence of dunes on dredging loads into account.

An additional clearance depth of 0,5 m was also used in the dredging settings to make sure that the navigation channel keeps its required depth at least for some time after the dredging effort. On the Parana River it is common practice to dump the dredged material to the side of the dredging location. Therefore, we used this as the reference dump strategy in our modelling study. Together with two alternative approaches the investigated dump strategies were:

- Du0 - dump to the side (reference strategy)
- Du1 - dump at least 0.5 km downstream (to the side)
- Du2 - dump in the nearest location in secondary river channels, on the opposite side of the river islands Isla Fontana (near Paso Bella Vista) and Isla la Carlota (near Paso Borghi), see Figure 3.

The latter two strategies (Du1 and Du2) were chosen to investigate whether moving bed material further away from dredging sites may eventually help in decreasing the dredging volumes.

Results

Cumulative dredging loads along the simulated reach and over the six year period are shown in Figure 4a, and the corresponding total dredging loads per year are reported in Figure 4b. The two shallows, Bella Vista and Borghi, clearly show up in the model results as areas of high dredging activity, with most material being dredged at Bella Vista.

For the six year simulation period, dumping dredged material downstream (Du1) reduced total dredging loads by ~20%, as can be seen by comparing Dr050Du0 to Dr050Du1 in Figure 8. Probably, this dredge reduction is due to the avoidance of sediment returning to its original dredging location. Moving the dredged material to distant locations in the side channel (Du2) yielded an even larger reduction of in dredging

loads of ~40% (compare Dr050Du0 to Dr050Du2).

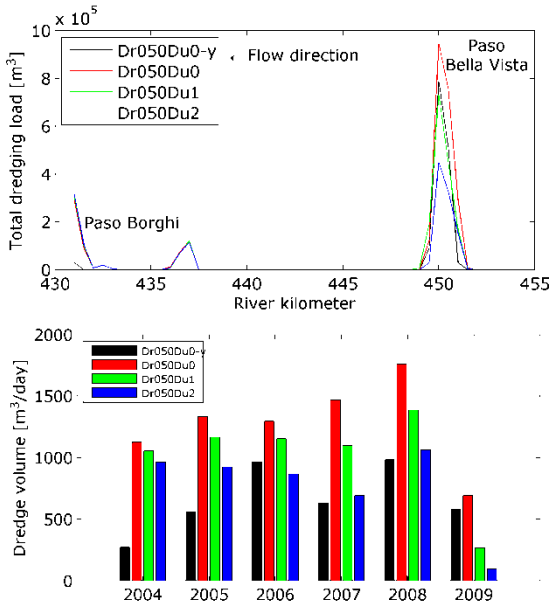

Figure 4. (a) Total dredging loads over the six year simulation period and (b) corresponding total dredging load for each scenario per year (for the reference dump strategy, the results for the "yearly Q" hydrograph were included that is Dr050Du0-y, as well).

By using the "stepped 10-day Q" hydrograph especially the low water levels trigger dredging; these low water levels are not taken into account using the "yearly Q" hydrograph. In 2006, the year-averaged discharge was relatively low. However, the dredging load for the "yearly Q" hydrograph was still lower than using the "stepped 10-day Q" hydrograph. We conclude that it was crucial to take into account the river's yearly discharge-variability, in particular the low flows, in order to properly represent dredging activities as they are performed in reality.

Moving the dredged material to distant locations in the side channel reduced the dredging loads in the navigation channel significantly. This dredge-and-dump strategy not only avoids return of material to its dredging location, it also affects the discharge distribution between the main channel and the side channels. Figure 5a shows that at Bella Vista, on the long run, the dump-strategies where the sediment was dumped near the main channel (Du0 and Du1), resulted in a gradually increasing fraction of the total discharge passing through the secondary channel.

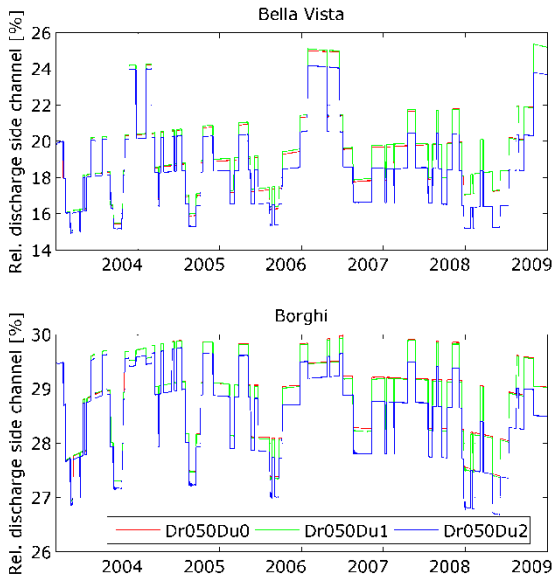

Figure 5: Time-series of relative discharge (a) in the secondary channel near Bella Vista, and (b) in the secondary channel near Borghi.

In contrast, in the dump-strategy where sediment was moved further away towards the side channels (Du2), the discharge distribution between the main channel and secondary channel was kept more stable over time. After a period of six years, at Paso Bella Vista even 2% additional flow is redirected to the navigation channel in the river main channel. At Paso Borghi this effect is also present (Figure 5b), but the effect is smaller (~0.75%) because less dredging (and thus dumping) takes place at this location. These hydrodynamic effects contributed to lowering sedimentation of the main channel and thus reduced necessary dredging loads.

Discussion

The study showed that the largest reduction in total dredging load could be achieved by dumping far away from the dredging location, in our case in the secondary channels on the opposite side of local river islands. This strategy gave two beneficial effects: (i) it avoided circular dredging, because the dredged material was moved to locations far away, and therefore material was not transported back into the dredging zone; and (ii) dumping the material in the side channel redirected more flow to the main channel of the river where the navigation channel is located and, hence, lowers sedimentation effects in the main channel. By dumping material in the more distant side channel the flow in the river was directed to be beneficial for river navigation, and in the long term could even further reduce dredging loads. On the other hand, this

dredging strategy could eventually lead to closing of the side channel, setting off morphological effects that may be harmful, for navigation but possibly also for flood risk or environmental aspects.

Conclusion

A computational morphodynamic study was carried out to explore the feedbacks between dredging-and-dumping strategies and river morphology in the medium-long term. Such studies may assist sediment management and the optimization of dredging-and-dumping strategies in river navigation channels where regular and significant maintenance activities are required. For the considered case of the Parana River (Argentina) we found that it is crucial to take into account the river's discharge-variability in order to properly represent dredging activities in the model.

Also, we found that strategic dumping of dredged material can significantly reduce total dredging loads, by avoiding redistribution of dredged material to shallow regions and by steering hydro-morphodynamic feedback processes that lower sedimentation at critical shallow zones. It was shown that moving dredged material downstream (by 500 m) as opposed to sideway-dumping could reduce total dredging loads by ~20%, because it avoids that sediment returns to its original dredging location. Even larger benefits were achieved by moving the dredged material to distant locations in the secondary channels, yielding ~40% reduction in dredging loads. In the considered case the moved material was not only safe from returning to its dredged shallow zones in the navigation channel, the dumped material also helped to stimulate flow dynamics in the river that aided maintenance of a deep enough, and more stable navigation channel. Both effects helped reducing dredging loads, leading to important reductions already starting in the year of implementation of the strategy and leading to even larger reductions in subsequent years.

The investigated dredge-and-dump strategies explored in the current study are not limited to Parana River, but can also help reduce dredging activities in navigation channels elsewhere around the world. Studies as these should become standard practice to aid navigation channel maintenance in rivers worldwide, not only to help reduce dredging costs, but also to harmonize natural river processes with navigation needs.

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Challenges in morphodynamic laboratory experiments

F.E. Roelvink[1,*], A. Lako[1,*], T.B. Le[1,2], A. Crosato[1,3]
[1]Delft University of Technology, PO Box 5048, 2600 GA Delft, the Netherlands
[2] Water Resources University, 175 Tay Son, Dong Da, Hanoi, Vietnam
[3] UNESCO-IHE, Department of Water Engineering, PO Box 3015, 2601 GA, Delft, the Netherlands
** Corresponding author; e-mail: anouklako@hotmail.com*
** Corresponding author; e-mail: floortje.roelvink@hotmail.com*

Introduction

The formation of bars, large sediment deposits, is of great hinder to the navigability of a river, as they decrease the local depth. Until now groynes are built to prevent bar formation and increase the water depth by reducing the channel width. Along the well-trained Waal River, Dutch authorities are considering substituting the old groynes with longitudinal training walls (Eerden, 2011). The goal is to obtain an improved navigation channel, while preserving the river conveyance capacity during floods, and improve the river bank natural value. However, the side-effects of the training wall must be examined with great care. For this we carried out laboratory experiments in a straight flume at the Civil Engineering and Geoscience faculty of TU Delft to study: 1) the morphological response at the reach scale (Lako, 2015), 2) the sediment- and discharge distribution at the channel partition that is induced by a training wall (Roelvink, 2015) and 3) the importance of the position of starting point of the longitudinal training wall (Le, 2015). During the experiments we were confronted with many challenges which made it difficult to obtain univocal results. This paper gives an overview of the challenges that flume experiments bring when simulating the river morphodynamics in order to improve the accuracy and reliability of future experiments, and thus enhance our knowledge of river management.

Figure 1: Alternate bars in Japan, Source: Vincent Langlois 2007-14

Method

The flume was filled with sand with a median grain size of 0,370 mm. The position of the training wall was determined using the results of a reference case, which showed the locations of the alternate bars formed without training wall. Two scenarios, with the training wall on different locations relative to the location of the bar were investigated, see Figure 5.

The instruments used for measuring were two discharge meters, a laser device for the bed- and water level measurement and a high speed camera for the discharge distribution. For every set of experiments, the measurements were carried out three times per time step.

The following quantities were measured: the bed –and water level, the flow velocity and the sediment transport velocity.

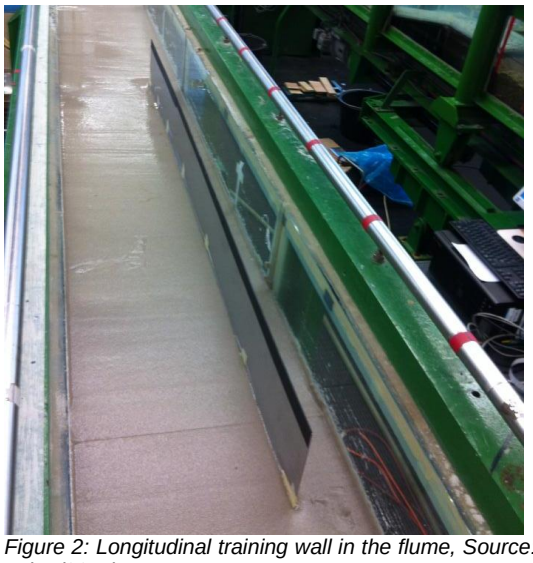

Figure 2: Longitudinal training wall in the flume, Source: A. Lako (2015)

Challenges

Obtaining a system with alternate bars

In order to obtain a system with alternate bars, the formula of Crosato-Mosselman (2009) for the most probable bar formation is used:

$$m^2 = 0{,}17g\frac{(b-3)}{\sqrt{\Delta D_{50}}}\frac{B^3 i}{CQ} \quad (1)$$

In which: m = most probable bar mode (-), g = acceleration due to gravity (m/s^2), b = exponent of the flow velocity in the sediment transport law, B = channel width (m), ΔD_{50}= sediment median grain size (m), C = Chézy coefficient, and Q_w= representative water discharge (m^3/s). Obtaining bar mode m=1 (alternate bars) poses several challenges, the most important one being the fact that the correct discharge has to be selected, corresponding to a representative Chézy coefficient. To select the right discharge, a relation between the Chézy coefficient and the discharge is used:

$$C = \frac{Q_w}{\sqrt{hi}Bh} \quad (2)$$

In which: C = Chézy coefficient, Q = discharge, h=water depth, i= bed slope, and B = channel width. With preliminary experiments and an estimation of the Chézy coefficient to be between 15 and 30, a certain preliminary value for the discharge is calculated, and with this discharge the experiment is started. The water depth is measured and together with the discharge this gives a value for Chézy and for the bar mode, according to the formulas above.

Still, this is a very rough approach to calculate the roughness coefficient, since the Chézy coefficient varies with varying bed configuration during the experiments, which means that the selected value for the discharge is only an approximation. Another problem is that it can take a long time before alternate bars are formed. This problem is, however, easily solved by placing a groyne at the upstream part of the flume, which speeds up the bar formation process and brings its duration back to 2 to 3 days.

Avoiding ripples on the flume bed

Formation of ripples on the flume bed is undesirable, since they influence the roughness of the flume bed and therefore the Chézy coefficient. Moreover, when the ripples are larger than the bars, it's difficult to distinguish ripples from bars. During the experiments of the research by Roelvink (2015) and Lako (2015), where sand with median grain size of 0,370 mm was used, this was the case. This was not convenient for the results, but by making use of the difference in wavelength between the ripples and bars, these could largely be filtered out. It was only after these experiments that Le (2015) found a better type of sand, which avoided ripple formation on the flume bed due to a larger division in grain size distribution. This makes it easier to estimate the Chézy coefficient and therefore makes the measurements more reliable.

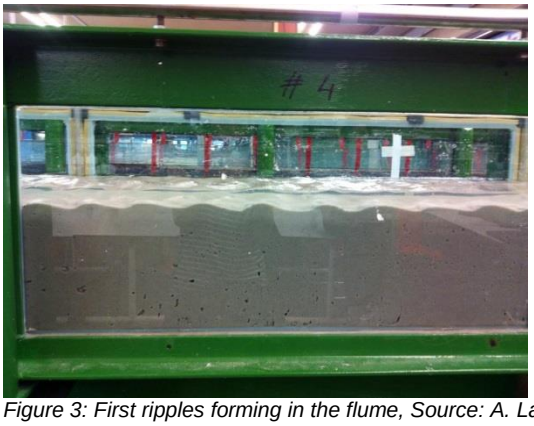

Figure 3: First ripples forming in the flume, Source: A. Lako (2015)

Measuring the discharge

The discharge was measured in three ways. The inflowing discharge was measured with both a Rehbock meter and a meter that was installed on the pump. These instruments are not very accurate, making it difficult to assess the actual amount of water flowing through the flume. To calculate the discharge distribution, which can be done by obtaining the discharge in both main -and side channel, a high speed camera was used in combination with black floating particles that were released upstream of the channel partition. The data were processed with PTV-lab, an application of Matlab that calculates the velocity of particles from video frames. With this, the surface velocity was obtained. However, this velocity was not equal to the median velocity in a cross-section, as there is a vertical velocity distribution. To obtain this median velocity, an empirical multiplication factor of 0.85 was used. Further, to calculate the discharge, the cross-sectional area was needed. This induces a second inaccuracy, as there are only four lasers to measure the water depth, which means an average value for the water depth in a cross-section needs to be used, further increasing the inaccuracy. This was confirmed by the fact that the sum of the discharge in the main and the side channel that was computed with PTV-lab, was not equal to the inflowing discharge that was measured with the Rehbock meter. A suggestion to solve this is to calibrate the sum of the discharge in the main and side channel to the inflowing discharge that is measured with the Rehbockweir meter. In order to do this, the Rehbockweir meter must be watched carefully, as the pressure in

the pump can change during the experiment as a result of the turning on and off of other flumes.

Measuring the sediment transport

Various optical and acoustic instruments exist to measure the sediment transport. However, these techniques are not broadly available, also not at TU Delft. The first idea was to track the movement of colored particles with the high speed camera and process these videos with PTV lab. However, because of the reflection of the water and the small size of the sediment, this was not possible. For this reason, the sediment transport was computed using the results of the bed and water level measurements. In this method, the transport rate of the ripples was taken as a measure for the sediment transport (Roelvink, 2015). The movement of the ripples was marked, and together with the elapsed time, a certain velocity could be assigned to every ripple. This technique enables us to give an estimate of the order of magnitude, but a comparison of the results showed a high distribution in values. This has to do with a couple of factors. Firstly, the transverse bed movement is not taken into account and neither is the transport of suspended load that floats through the water. Secondly, sediment rolling over the ripples is neglected. An interesting point to note is that the ripples in this experiment are an unwanted phenomenon that is caused by the choice of sediment size and the scaling down from a real river to the flume. So if possible, ripples should be avoided. However, without ripples, this technique cannot be applied anymore, making this a point of consideration.

Measuring the bed-and water level

Firstly, the measuring equipment used during the experiments all have a certain accuracy. For the case of flume experiments in particular, it is important to be aware of the noise in the data and the different factors that cause it. Since the laser device measures by reflection of opaque matter, jumping sediment causes noise in the bed level measurements. The water level can be measured by any kind of floating object on the water surface that can reflect the laser, but note that this will likely cause ripples in the water and therefore noise. By knowing the causes of the noise, one can minimize these effects. Additionally, large deviations can be filtered out by an appropriate Matlab program. Another thing that one should not forget is the different refractive index of water, when converting the laser output (in Volt) into distance (in mm). This refractive index should be used when calculating the real bed and water level by subtracting the values measured by the laser from the distance between laser and bottom of the flume.

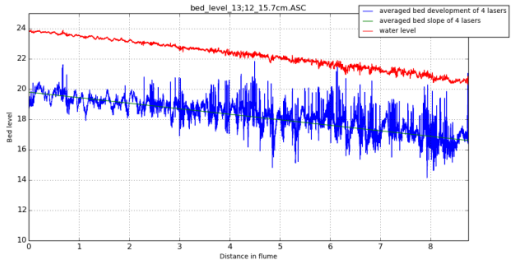

Figure 4: Noise in the laser data. Source: A. Lako (2015)

Establishing morphodynamic equilibrium

In order to assess the long-term response of rivers to human interventions, the equilibrium theory can be used. This theory is based on a comparison of two morphodynamic equilibrium states: one before an intervention and one a long time after the intervention. This means that at some time after the intervention, the river has adjusted its width, depth and surface slope such that a new morphological equilibrium is reached. The challenge here is to make sure that morphodynamic equilibrium is actually reached. This means that the reach-averaged characteristics (longitudinal bed slope, water depth) do not change anymore, so that has to be checked carefully. Moreover, the flume has to be kept in equilibrium, for which sediment input has to be equal to sediment output. This is another challenge, which is discussed below.

The in –and output of sediment

According to the equilibrium theory, sediment input has to be equal to sediment output in a certain control volume. Considering that sediment is leaving the flume via the weir at the end to the return gutter, this is not naturally the case. Therefore sediment would have to be either recirculated or continuously fed, in order to achieve equilibrium. Since there was no sediment recirculation system during the research by Roelvink (2015) and Lako (2015), and continuously feeding of sediment requires continuous presence and precise dosing of sediment, none of these two techniques was used. Instead, extra sediment was inserted at the beginning of the flume, assuming this would be enough to supply the flume with enough sediment for the duration of the experiments. This however, turned out to be a wrong assumption, since a high rate of scour was observed near the groyne, so sediment output was higher than sediment input and equilibrium was most likely disturbed. From this we can conclude that for future research, sediment recirculation is indeed necessary.

This also brings new challenges, like matching the sediment input with the sediment output.

Establishing the position of the training wall

Using the results of the reference case, the position of the wall is determined. This reference case is the case without any perturbation of the flow, so without the longitudinal training wall. Two scenarios are investigated, one where the wall results in the silting up of the side channel and one where the side channel quickly erodes. For this, it is very important to know the exact location of the upstream end of the training on a bar. Regrettably, the location of the bars can't be defined that strictly as there is a variation in locations between different reference cases and because the bars migrate during the experiment. Therefore, one must be hesitant in making conclusions about the effect of the training wall. Further, as the experiment was running, sediment left the flume at the downstream end, resulting in a lowered bed level. This influenced the development of the side channel, resulting in an area of sedimentation that is hard to recognize.

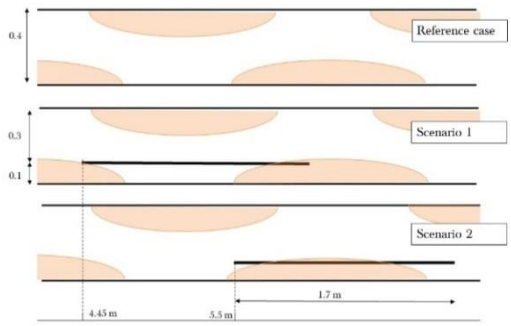

Figure 5: Three scenarios of the training wall, Source: F.E. Roelvink (2015)

Scale effects

Conducting flume experiments on a small scale means that spatial and temporal scale effects play a role too. This causes local perturbations such as scour near the upstream end of the training wall, which does not match the scour that would be seen in the real river. This cannot be prevented but it is something that has to be taken into account.

Temporal scale effects; long term response of a river to a longitudinal wall

Bars can be either migrating or steady. At first, migrating bars dominate the flume bed, but as the experiment runs, the wavelength of the bars increases and steady bars start to form. These steady bars dominate the bed of alluvial rivers that are subject to local perturbations (Crosato & Mosselman). In the flume, steady bars occur as a result of the groyne that is placed near the entrance of the flume. However, these bars are steady at a certain time scale and are in fact still moving, albeit very slowly. This makes it difficult to assess the influence of the wall on the river morphology in the long term. To do this, the experiment should be kept running for much longer, to see if the longitudinal wall is a stable solution or that its impact changes in time.

Conclusion

During the research on the morphological response on the reach-scale by Lako (2015) and sediment- and discharge distribution by Roelvink (2015) many challenges were encountered during the morphodynamic laboratory experiments. Most challenges were related to the attempt of reproducing a real river at a much smaller scale: establishing a system with alternate bars, as well as establishing equilibrium. Other challenges were related to the measurements. The measurement of bed- and water level proved to be fairly easy, whereas measuring the sediment distribution was and is a big challenge. The research led to preliminary results and many recommendations that can be used for future research.

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2 – Presentations day 2

Turning the tide: unravelling biomorphodynamics of estuaries

M.G. Kleinhans
Universiteit Utrecht, Faculty of Geosciences, PObox 80115, 3508 TC Utrecht, the Netherlands
Email: m.g.kleinhans@uu.nl

Holocene estuary development

Estuaries are shaped by biomorphological processes, with patterns of channels and shoals, sand/mud flats, tidal marshes, vegetated banks and peat. Development was influenced by early Holocene landscape that drowned under sea-level rise, and by human interference. Estuaries harbour highly productive natural habitats and are of pivotal economic importance for food production, access to harbours and urban safety. Accelerating sea-level rise, changing river discharge and interference threaten these functions, but we lack fundamental understanding and models to predict combined effects of biomorphological interactions, inherited landscape and changing drivers.

We do not understand to what extent present estuary planform shape and shoal patterns resulted from biomorphological processes interacting with inherited conditions and interference. Ecology suggests dominant effects of flow-resisting and sediment de/stabilising eco-engineering species. Yet abiotic physics-based models reproduce channel-shoal patterns surprisingly well, but must assume a fixed planform estuary shape. Holocene reconstructions emphasise inherited landscape- and agricultural effects on this planform shape, yet fossil shells and peat also imply eco-engineering effects.

Our aims are to develop models for large-scale planform shape and size of sandy estuaries and predict past and future, large-scale effects of biomorphological interactions and inherited conditions.

We will combine state-of-the-art eco-morphological model, unique analogue landscape models with eco-engineers and a new, automated palaeogeographic reconstruction of ten data-rich Holocene estuaries on the south-east North Sea coast. We will systematically compare these to modelled scenarios with biomorphological processes, historic interference and inherited valley geometry and substrate.

Decadal channel-shoal interactions

Tidal systems such as the Scheldt, Humber and Columbia estuaries and Wadden seas in Florida and the North Sea, have perpetually changing and interacting channels and shoals formed by ebb and flood currents. Current models fail to forecast these natural dynamics. Yet main channels are economically important shipping fairways, whilst shoal areas that emerge and submerge daily are ecologically valuable habitats under threat of dredging, dumping and sea level rise. We urgently need dynamic forecasting models to optimise management strategies for these multiple functions.

Our aim is to investigate and forecast how channel-shoal dynamics in estuaries result from geomorphological processes and human interference.

I hypothesise that channel- and shoal-margin collapses and current-driven sand transport on sloping channel beds cause the dynamics of channels and shoals, whilst break-down of shoals is balanced by resistant cohesive mud layers. The bifurcating channel network propagates and possibly amplifies small-scale disturbances by collapses and dredging through the system into neighbouring reaches.

We are commencing complementary experiments and numerical modelling: we create experimental scale models in a unique tidal laboratory facility based on a pilot setup that is the first ever to form sustained-dynamic tidal channels and shoals. We are developing three critical model components for a state-of-the-art numerical model: 1) sand transport on gentle channel slopes, based on recent theory and new rotating-flume experiments, 2) mud layer formation, based on recent model advances, and 3) large bank collapses, based on geotechnical modelling and historical data.

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Analyzing collaborative relationships regarding floodplain management through social network analysis: a Dutch case study.

J.M. Fliervoet*[1], G.W. Geerling[12], E. Mostert[3] & A.J.M. Smits[1]
[1] Radboud University Nijmegen, Institute for Science, Innovation and Society, P.O. Box 9010, 6500 GL Nijmegen, The Netherlands
[2] Deltares, P.O. Box 177, 2600 MH Delft, The Netherlands
[3] Delft University of Technology, Stevinweg 1, 2628 CN Delft, The Netherlands
** Corresponding author; e-mail: j.fliervoet@fnwi.ru.nl*

Introduction

As 'Room for the River', will come to a close by the end of 2015, river management in the Netherlands is shifting towards maintaining multi-functional floodplains. This implies a new focus for the current collaborative processes between public and private stakeholders and brings new collaborative challenges.

Until recently, public organizations played a dominant and decisive role in the water sector. However, an increasing number of studies emphasize the shift towards partnerships and collaborative governance. In other words, the dominant role of the public organizations is developing towards a more facilitating role as equal partner to improve efficiency and create a leaner state. In the literature this shift is expressed in the catchphrase *"from government to governance"* (Huitema and Meijerink, 2014; Mostert, 2015; Rhodes, 1997; Termeer, 2009)

Advantages of collaboration are increasing public support, reduce opposition and improve implementation of government policy. Other reasons given to collaborate include the dependency on other stakeholder, because of limited resources of government: government simply does not have all the information, power and finances necessary for environmental management (Gray, 1989; Huxham and Vangen, 2005). Budget cuts over the past years have only increased this dependence. Additionally, studies refer to the moral argument of involving diverse stakeholders to make environmental management more democratic (Mostert et al., 2007; Stringer et al., 2006).

Despite these reasons given to collaborate Watson et al. (2009) and Benson et al. (2013) argued if this really made environmental management more collaborative? To understand the current and complex collaborative relationships in the field of river management, we conducted a social network analysis of floodplain management in the Dutch Rhine delta. Social networks analysis consist of a set of nodes (in this case organizations) and a set of ties between these nodes (collaborative relationship).

The multi-functional design of the Dutch floodplains included diverse stakeholders, which created interdependence to the different functions, especially regarding flood protection and nature restoration (Fliervoet et al., 2013). Therefore, we analyze both the "blue network" concerning flood protection and the "green network" concerning. The following questions will be addressed:

(i) Which actors are involved and what are their collaborative relationships to ensure flood protection (blue network) and/or reach nature objectives (green network)?
(ii) Which actors play a coordinating or bridging role?
(iii) What is the role of governmental versus non-governmental organizations in both networks?

Briefly, based on an actor-centered approach this article explores how transboundary river management is designed and what opportunities (on the level of "structures") can be identified.

Method

The collaborative activities concerning the maintenance of the floodplains along the River Waal are central to this case study. The boundaries are defined by the provincial program of *'WaalWeelde'* – from the municipality of *Rijnwaarden* to the municipality of *Zaltbommel* (a distance of approximately 80 km on both sides of the river). This program connected public, private and societal organizations in the planning and implementation phase of multifunctional river management along the intensely navigated River Waal (Smits 2009). Based on a bottom-up approach, this integrated multi-player program aimed to develop a safer, more natural and economically stronger riverine landscape.

A quantitative research was conducted based on a survey. The survey consisted of a list of seventy organizations, including

governmental organizations, non-governmental organizations, businesses, knowledge institutions and associations of farmers. The respondents were asked to select from the list of 70 organizations, the organizations with which they interacted and to indicate the strength (frequency) of their interactions, for flood protection and nature objectives separately. This resulted in a response rate of 73 percent.

Social Network Analysis (SNA)

The survey data of 43 actors was analyzed regarding collaborative ties by using the software program UCINET (Borgatti et al., 2002). For the SNA we used primarily reciprocated collaborative ties, meaning that both actors indicated that they collaborated. Since each tie depends on two actors, the data is more robust to reporting errors (Stein et al., 2011).

The networks were analyzed at three levels, i.e. (1) the network as a whole, (2) actor-groups and (3) individual actors, by using metrics like density, group exchange, degree centrality and betweenness centrality. The metric density divides the number of ties in the network by the maximum possible number of ties. The reciprocal ties connected to one stakeholder group divided by the total number of reciprocal ties in the network refers to the group exchange. On the individual level, the degree and betweenness scores show respectively the number of ties an actor possess and the probability of an organization being the shortest path between any two organizations in the network. Both metrics provide insight in the position and role of individual actors in the network and help to identify central, coordinating and bridging organizations whose activities connect actors that otherwise would not have been connected (Rathwell and Peterson, 2012).

Results

The whole-network characteristics indicated well connected and heterogeneous networks when focusing on all collaborative frequencies (Yearly, quarterly, monthly and weekly). The blue network consisted of 36 actors with reciprocal ties (out of 43 in total), and green network of 42 actors. The density of the green network is 30 percent higher in contrast to the blue network, even with the higher number of actors.

A large decrease in the connectedness is found for both networks, when focusing on two highest tie-strength classes (monthly and weekly). Twelve organizations drop out of the blue and green network on top of the already disconnected actors, especially actors of the research, agriculture and special interest group (Figure 1). Figure 1 indicates the few ties between the nature (group 4) and flood protection organizations (group 1) for both networks. Moreover, it shows the central position and importance of Crd1, the Government Service for Land and Water Management, in the collaborative networks.

When we distinguish between governmental and non-governmental actors, we can clearly see the importance of the former (gray nodes in Figure 1). Governmental organizations account for 75 and 65 percent of all ties (monthly and weekly collaboration) for respectively the blue and green network. The actor groups were grouped by their main tasks and function. The nature group possess 36 percent of the ties in the green network.

In both networks, the central position is occupied by Crd1, based on the highest degree and betweenness scores. This actor is a major broker between the nature and flood protection organizations, but also among the group of coordinators of spatial planning (group 6 in Figure 1). In the green network the second place is held by the State Forestry Service (Nat7) and in the blue network by the Delta program (Fld1) based on reciprocal ties. Surprisingly, given their mandate, the provincial government (Nat6) and State Water Agency (Fld3) are not in the top 10 of reciprocal ties in both networks.

Discussion & conclusion

The social network analysis has shown complex and well-connected networks concerning the maintenance of flood protection (blue) an nature (green) objectives in the Dutch floodplains. According to Sandström and Carlsson (2008), the high connectedness promotes the potential for collective action and collaboration, especially when many ties exist between different types of actors (e.g. between recreational fishermen and governmental officials). A disadvantage of dense networks is the obstruction of perceptions outside the networks, this may reduce the capacity to innovate.

In the beginning of 2015 the most central actor (Crd1), Government Service for Land and Water Management, was disbanded due to national state budget cuts.The discontinuation of this actor will decrease the connectedness of actors within the blue and green network and may therefore have a large impact on the exchange of ideas and decision-making processes.

Furthermore, our research shows the dependence of non-governmental actors on

the main governmental organizations. It seems that the Dutch governmental organizations still have a dominant and controlling role in floodplain management. This challenges the alleged shift from a dominant government towards collaborative governance and calls for detailed analysis of actual governance.

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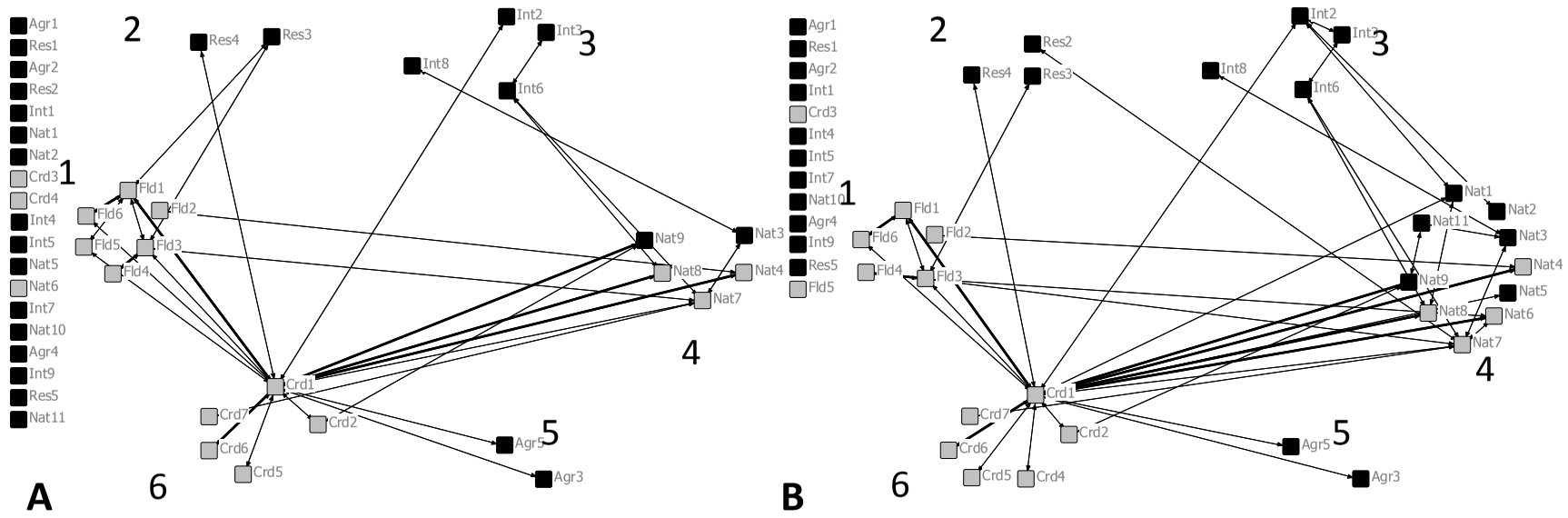

*Figure 1. Social network based on the **monthly and weekly** reciprocal ties concerning flood protection objectives (A) and nature objectives (B). Bold lines indicate the weekly ties. Grey nodes indicate a governmental organization and black nodes a non-governmental organization. Numbers indicate the interest of the six groups: (1) flood protection; (2) research institutes; (3) special interest groups; (4) nature; (5) agriculture; and (6) coordination or spatial planning.*

Predicting concentrations of human pharmaceuticals throughout the river systems of Europe

A.C. Johnson[a], R. Oldenkamp[b], E. Dumont[a], R.J. Williams[a], I. Cisowska[a], J.P. Sumpter[c]
[a] *Centre for Ecology and Hydrology, Wallingford, Oxfordshire, United Kingdom*
[b] *Department of Environmental Science, Radboud University, Nijmegen, The Netherlands*
[c] *Institute for the Environment, Brunel University, Uxbridge, United Kingdom*

Introduction

Human pharmaceuticals are produced in significant amounts with high production volume. Not surprisingly, pharmaceuticals have been detected around the world in a wide range of environmental media, such as urban and hospital wastewater effluents, surface waters, ground waters and drinking waters (e.g., Fatta-Kassinos et al., 2011). Consequently, the discharge of pharmaceuticals in wastewater into the aquatic environment has been a source of discussion and concern in scientific and regulatory circles for more than a decade (Daughton & Ternes, 1999; Halling-Sørensen et al., 1998). The control of what are called hazardous substances in Europe falls under the Water Framework Directive (WFD). When an environmental quality standard (EQS) is set for a chemical this can lead to it being phased out of production. However, the recent addition of the pharmaceuticals 17α-ethinylestradiol (EE2), 17β-estradiol (E2) and diclofenac in the European Community document (COM(2011)876) appear to usher in a new era. The document suggested annual average EQS of 0.035 ng/L for EE2, 0.4 ng/L for E2 and 100 ng/L for diclofenac.

If these three pharmaceuticals stay on the watch list and even become priority substances needing control, so it is likely other pharmaceuticals will follow. Chemotherapy drugs in the group known as cytostatic, cytotoxic, or antineoplastic (referred to collectively as cytostatic) are often featured on lists of pharmaceuticals of concern that are discharged into our river systems (Fent et al., 2006; Sanderson et al., 2004), due to their possess fetotoxic, genotoxic and teratogenic properties, which have already been shown in fish (Grisolia, 2002) and invertebrates (Anderson et al., 1995).

Given their societal health benefits, it is unlikely and perhaps undesirable for particular pharmaceuticals to be phased out on the basis of environmental concerns. Thus, as source controls are inappropriate, so end of pipe solutions may have to be sought. Geographic-based water quality models are a practical tool that can address the question of exposure to pharmaceuticals at a continental scale. Measuring all of these chemicals throughout every European river would be exceedingly costly and time consuming, to say nothing of the problems of consistency and technical feasibility.

Here, the results of two separate but related studies are presented, in which we attempted to predict the range of possible concentrations of pharmaceuticals in surface waters for various countries in the European Union. Based on publically available consumption data, literature data on human excretion values, and sewage removal rates, we predict concentrations of four cytostatics (i.e., cyclophosphamide; CP, carboplatin, 5-fluorouracil; 5FU, and capecitabine, as well as the three "WFD-compounds" E2, EE2, and, using the geographic-based Global Water Availability model (GWAVA; Dumont et al., 2012). For the latter three compounds, we examined whether and where predicted river concentrations would exceed proposed EQS levels of 0.4 ng/L for E2, 0.035 ng/L for EE2 and 100 ng/L across Europe.

Method

Estimation of effluent concentrations

The approach to estimating sewage effluent concentrations takes the drug consumption per capita for a specific country, less that prevented from being excreted as the free parent compound, and less that removed in sewage treatment. The effluent concentration is then calculated by dividing this figure by the per capita wastewater discharge for that nation (Equation 1).

$$W = \frac{(C-E-S)}{D} \qquad (1)$$

Where *C* is consumption of the drug (ng/cap/d); *E* is the amount of the drug that is not excreted (ng/cap/d); *S* is the amount of the drug that is prevented from escaping into sewage effluent (ng/cap/d); *D* is the diluting volume of wastewater (L/cap/d); and *W* is the effluent concentration (ng/L).

The river concentration at the point of the effluent discharge (R_m, ng/L) is calculated by mass balance, and loss of the compound due to aquatic processes such as sedimentation and transformation is accounted for with a first order dissipation process to give the downstream concentration (R_d, ng/L) (Equation 2).

$$R_d = R_m \cdot e^{-k \cdot i} \qquad (2)$$

Where k is the decay rate (days^{-1}) and t is the time of travel (days). The time of travel is the river reach volume divided by the flow rate (Dumont et al., 2012).

Assessing consumption, excretion and sewage removal

The most critical part of any predictive model to assess concentrations of human derived chemicals in water is obtaining information on usage. Fortunately, some national annual consumption data on cytostatic drugs are publically available. These were interrogated to assess a per capita consumption value, given the population of the country at a particular time.

Next, the extent to which the parent compound is excreted unchanged by the patient has to be considered. Not surprisingly, humans vary in their excretion behaviour, with such factors as age, health, and co-medication all influencing the percentage excreted. This especially holds for the cytostatic compounds assessed. We therefore surveyed a wide range of literature on excretion rates before arriving at a mean value. Similarly to excretion rates, natural variations in sewage performance can influence pharmaceutical removal rates in treatment. Moreover, the literature on removal in sewage treatment for many (cytostatic) pharmaceuticals is still limited.

European river water modelling

To examine potential concentrations of the pharmaceuticals throughout European surface waters, the geographic-based water resources model GWAVA was used (Dumont et al., 2012). This model uses geographic data on the location and size of the human European population and their association with sewage treatment plants (STPs) (EEA, 2011). The flows through these STPs are incorporated with the natural river discharge adjusted for abstractions (principally for potable supply and agriculture). The hydrology is driven by monthly climate over the period 1970-2000. The model calculates the water concentrations of chemicals through water courses in a series of 177,470 grid squares (cells) of approximately 6 x 9 km. On a monthly basis, in the water courses in each cell receiving effluent, the concentration is calculated by diluting the mass of chemical discharged in the volume of water in the cell, accounting for any loads from upstream cells. The chemicals are transported downstream with the discharge to the next cell. Chemical can be lost through abstraction or a first-order dissipation process (Equation 2). The time of travel through the gridded network (which can comprise rivers, lakes and wetlands) is calculated from the river flow rate and the water volume of each cell. Surface water volumes are estimated using established empirical relationships with width and depth data (Dumont et al., 2012).

Results

The GWAVA model provides predictions for 1.2 million km of European rivers receiving the waste from 602.8 million people. As such, a single run of the model with its 177 000 grid squares and 31 years of climate data generates 66 million results per chemical. All of the variables discussed will play a role. However, the most important factor in correctly predicting river concentrations, apart from consumption, is dilution. Different interpretations on human excretion, or sewage removal rates, could change the values by up to 20-fold, but dilution could change the values by up to 1000-fold.

Predictions for the four cytostatics

The results from model runs are displayed in a map showing the 50th percentile concentrations across Europe for CP based on a mean excretion rate and mean sewage treatment removal (Fig. 1). This is broadly equivalent to the concentration that would be recorded at a median flow for that part of a river and, as such, might represent the typical exposure for surface waters. When potential worst-case river concentrations are modelled, such as might be associated with low summer flows, the simulations indicate that 99% of European river locations would be below 0.2 ng/L for carboplatin and below 0.6 ng/L for 5FU. With CP, only 0.1% of locations could exceed 1 ng/L, whereas for capecitabine, 2.2% could exceed 1 ng/L in rivers.

Predicted exceedances of proposed European EQS values for EE2, E2 and diclofenac

Before starting the water quality modelling, based on the European mean consumption values, excretion values and sewage removal

factors, then 1 ng/L EE2, 3 ng/L E2, and 570 ng/L diclofenac would be expected in European sewage effluents. At mean excretion and mean sewage removal, rivers where an annual average concentration of EE2 would exceed 0.035 ng/L would be fairly widespread with the expected scenario (Fig. 2). Of perhaps greater biological significance is where EE2 concentrations might exceed 0.35 ng/L (Caldwell et al., 2008), and this is far less widespread but not negligible (Fig. 2). When all the results are plotted as cumulative frequency distributions and compared with the proposed EQS values, it can be seen that EE2 would pose the greatest challenge (Fig. 3). Between 2 and 25% by length of Europe's rivers were predicted to have EE2 concentrations in excess of 0.035 ng/L (best and worst case) with the expected outcome being 12% (Fig. 4).

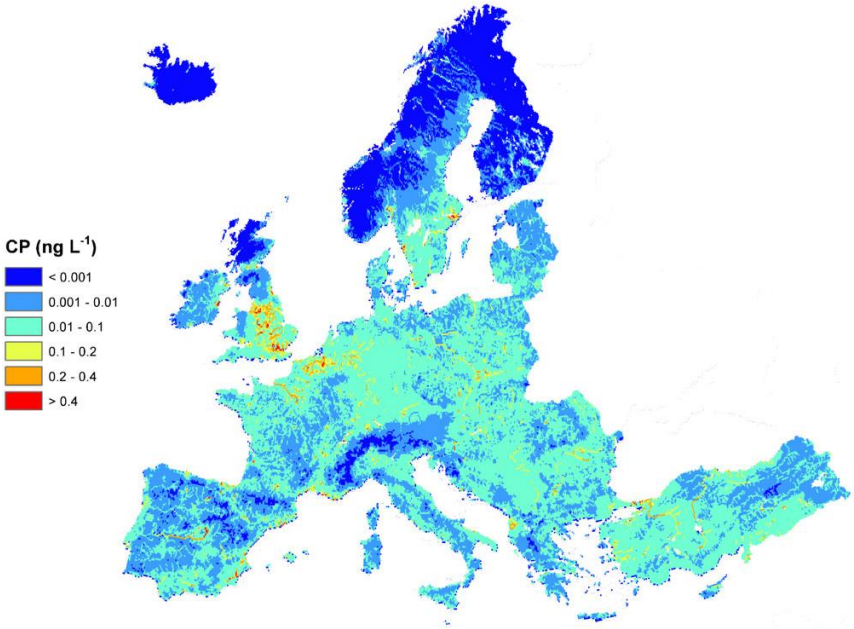

Figure 1. Predicted cyclophosphamide (CP) concentrations in surface water based on mean excretion rate, mean sewage treatment removal, and 50th percentile flow across the European continent, taking into account differing national per capita consumption and wastewater discharge values from the Global Water Availability model (GWAVA).

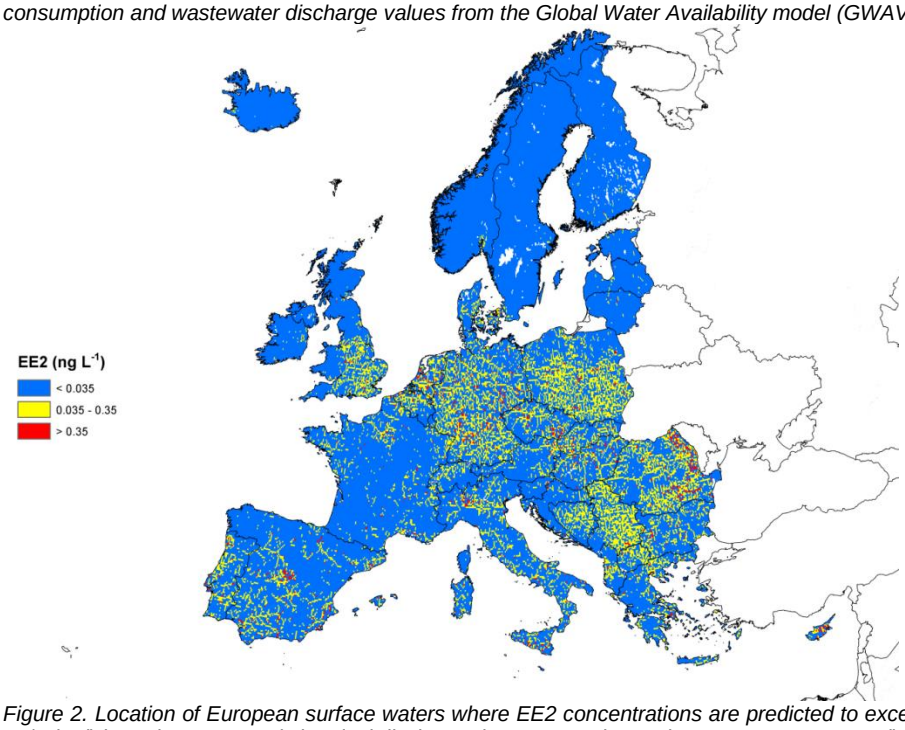

Figure 2. Location of European surface waters where EE2 concentrations are predicted to exceed 0.035 ng/L (yellow) and 0.35 ng/L (red) based on expected chemical discharge (mean excretion and mean sewage removal).

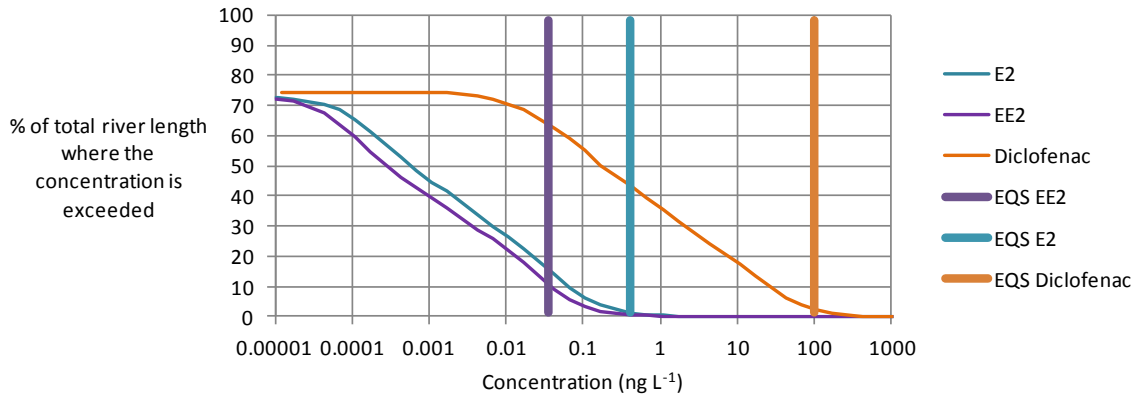

Figure 3. Predicted average river water concentrations throughout the European river network based on expected chemical discharge (mean excretion and mean sewage removal) and their proximity to the proposed EQS values in COM(2011)876.

Conclusion

Concentrations predicted for cytostatics

We found a surprising difference in the popularity of the four cytostatics drugs across European nations, with differences of up to 28-fold. The predicted mean effluent concentrations ranged from 2 ng/L to 40 ng/L for CP, from 0.8 ng/L to 2.5 ng/L for carboplatin, from 0.3 ng/L to 2.5 ng/L for 5FU, and from 8.5 ng/L to 87 ng/L for capecitabine. In the majority of cases, where data are available, it is possible to predict CP concentrations in sewage effluent to within an order of magnitude of that observed (data not shown here). By linking with the geographic-based water quality model, it is expected that the majority of European rivers would have concentrations below 1 ng/L for these cytostatics drugs.

Comparison with proposed EQS values and implications

Given the enormous difficulties in measuring pictogram concentrations of E2 and EE2 in rivers, currently our best hope in assessing exposures throughout Europe is through modelling. With its global scope, models like GWAVA can be applied to continents, such as Europe, to assess possible river concentrations of pollutants originating from the human population. Despite limitations with respect to the resolution (6 x 9 km grid cells) and limited national consumption data for some countries (for which a European mean had to be applied), the clear message from this modelling exercise was that over 10% of continental Europe's rivers would exceed the proposed EQS for EE2 of 0.035 ng/L. If such an EQS were to be applied across Europe, it would represent an enormous technical and financial challenge to meet, given the extent of likely failure predicted here.

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What do we know about the composition of Dutch river beds?

A.P. Wiersma
Deltares, dept. of Applied Geology and Geophysics, P.O. Box 177, 2600 MH Delft, The Netherlands

Abstract

Knowledge about the composition of the riverbed is essential for river management and restoration. The presence of peat or clay layers below or next to the river channel slows down erosion and prevents groundwater infiltration, while sandy substrate below the active channel deposits can lead to infiltration and erosion.

The Netherlands has the best known shallow subsurface in the world, and hundreds of thousands of boreholes and cone penetration tests are digitally available. Still, relatively little is known about the composition of the river beds; i.e. does the river incise into sandy deposits or is the river channel bounded by clay or peat deposits?

In general, the subsurface around the big rivers in the Netherlands is composed of deltaic sediments and consists of peat, floodplain clays, overbank loams, tidal clays and sands and sandy (river) channel deposits. In the eastern part of the Netherlands, the rivers incise into the coarse sandy fluvial deposits below. In the western part of the Netherlands, these coarse fluvial sands lie deeper and the river channel is interlocked between the complex deltaic sediments described above. In the middle reach, the river locally cuts through the deltaic sediments and is locally bounded by them. Where exactly the river incises into sandy formations below, and where it possibly makes contact with erosion-prone deltaic deposits is not known.

The lack of knowledge about the subsurface below rivers has a number of consequences for river management. For the stability of dikes, it is important to know the potential infiltration of river water in sandy aquifers below. For river bed erosion, it is important to know the composition around and below scour holes to predict the evolution and potential hazard to river banks. Also, the location of potential weaknesses below the river bed is important to be able to anticipate in an early stage to unwanted effects. Lastly, for realistically modeling sedimentation and erosion patterns in rivers, knowing the subsurface composition is essential.

In the past years, for several river branches in the Netherlands studies have been carried out to determine the river bed composition and related risks, commissioned by Rijkswaterstaat and water boards. For some branches, only the available boreholes were used, whereas for other branches additional acoustic geophysical techniques were deployed and/or additional boreholes were acquired. In addition, bathymetric data, historical maps and earlier research such as sand ribbon maps were used.

By combining the findings of these studies, we provide examples of how bathymetric data of rivers can be used to reinterpret the river history and subsurface composition around it, and vice versa how the subsurface composition interacts with the river. In addition, we demonstrate the known and unknown aspects of the subsurface of the Dutch rivers, and show examples of methods for mapping the true composition of the river beds.

Bed topography reconstruction of meandering alluvial rivers from scarce data through combination of physical model bathymetry and spatial interpolation methods

D.I. Zervakis[1,2,*]
[1]*Delft University of Technology, Jaffalaan 5 - 2628 BX Delft, The Netherlands*
[2]*Deltares, P.O. Box 177, 2600 MH Delft*
** Corresponding author: D.I.Zervakis@student.tudelft.nl*

Introduction

Today nearly half of the global population lives closer than 3 km to surface fresh water, mainly including rivers and streams. Furthermore, an economic lifeline exists between ports and the inland waterway network of rivers, whose natural behaviour may cause local bottlenecks in navigability. Therefore, monitoring of the river channel bed is of outmost importance. However, bathymetric river data often do not cover the whole channel bed, but are instead scarce and insufficient for its precise depiction. To mitigate this, interpolation methods are applied, which can be highly inaccurate when available data are not closely and regularly sampled. For a better prediction, computational models and the human expertise are often required, which demands a much higher consumption of time and resources.

In this proposal, the bathymetry prediction from a linearized physical model is used together with the results of spatial interpolation methods in order to acquire a better representation of the channel river bed when the sampled data is scarce. The study is limited to meandering alluvial river channels, which are the most common river planform style in populated areas. The full process of reconstructing the river bed is presented and special mention is made about the importance of considering the anisotropic nature of river bed morphology and transforming the datasets in a flow-oriented coordinate system.

Scope

The focus lies on the implementation and assessment of a method that couples a spatial interpolation technique with the impact of river bed morphology in order to acquire an acceptable prediction of the river bed when the sampled data is scarce. The term “scarce” applies to limited bathymetric data, insufficient to provide a good estimation by normal interpolation methods. That is usually single-beam echo sounder datasets, which form cross-sections or trackline/”zig-zag” patterns (Fig. 1).

Meandering alluvial rivers with mild curvature and width changes are explored. The sampled data is to be presented on a curvilinear grid in order to transform them to a flow-oriented coordinate system and perform interpolation.

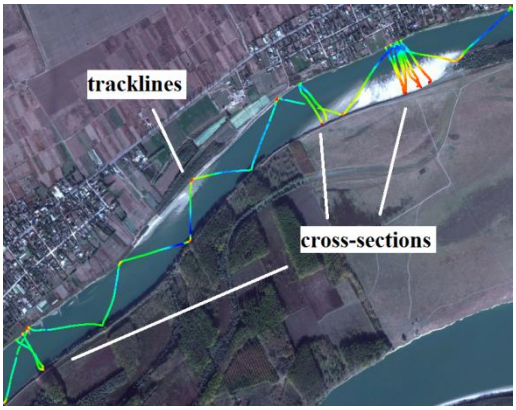

Figure 1. Limited (“scarce”) bed topography data.

The evaluation of the method is attributed to Root Mean Square Error (RMSE) and a visual assessment of error maps.

Related work

The background work is broken down to two main aspects involved in this project:

2. The linear physical model which predicts the river channel bathymetry based on a number of hydrographical parameters.
3. The spatial interpolation method and related considerations, like anisotropy and flow-oriented coordinate system.

Physical model

Various important past attempts have been made to describe the behaviour of river meanders (Ikeda et al., 1981, Struiksma et al., 1985). Modelling of the flow field has been attempted in (Kalkwijk & De Vriend, 1980), but for wide open channel bends of mild curvature. For such assumptions, the shallow water approximations hold, derived from the Navier-Stokes equations (Navier, 1822) and depth-averaged over a steady flow for a curvilinear system in *s* (longitudinal), *n* (transverse) and *z* (vertical) directions (Olesen, 1987) (Fig. 2).

In alluvial rivers sediment is transported but also deposited mostly on the inner side of bends, forming the so called point bars, which together with erosion on the outer bend, dictate the shape of cross-sections.

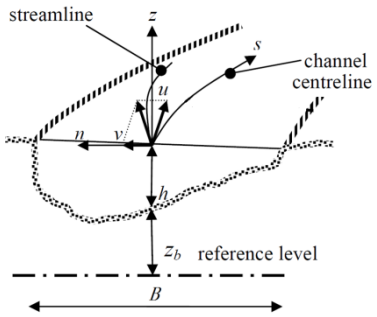

Figure 2. Cross-section of a river bend.

To support these profiles, a number of analytical equations have been derived to approximate the river bed topography based on the work by Crosato (2008).
A zero-order solution to the momentum and continuity equations provides the bed topography for a uniform flow in a straight and infinitely long channel, as follows:

$$h_c = \left(\frac{Q}{BC\sqrt{i}} \right)^{2/3} \qquad (1)$$

Where:
- h_c – water depth at the centerline [m]
- Q – bankfull discharge [m^3/s]
- B – river width [m]
- C – the Chézy roughness [$m^{1/2}$/s]
- i – the river (longitudinal) slope [-]

The way river bends can impact the bed level, is estimated by an axi-symmetric solution:

$$h_{(n)} = h_c e^{Af_{(\theta)} n / R_c} \qquad (2)$$

Where:
- h – water depth along n direction [m]
- A – coefficient (influence of the helical flow)
- f(θ) – weighing function based on Shields parameter (θ)
- n – coordinate orthogonal to streamline [m]
- R_c - radius of curvature [m]

Interpolation methods and spatial aspects
In the field of river research, a large number of spatial interpolation techniques are already extensively explored like Inverse Distance Weighting (IDW) (Philip & Watson, 1982), Kriging (Oliver, 1990), Natural Neighbor (Watson & Philip 1987) and Spline methods (Franke, 1982). A simpler solution is often accounted for by (bi-)linear interpolation which can be substantive for covering specific unsampled areas.
To narrow down the interest to the case of scarce data, geostatistical methods like Kriging are not considered. Their predictions are compromised by too far apart measured points that seem to have a greater correlation than closer ones because of the shape of the river bed. Also, Natural Neighbour tends to over-generalize the output especially in cases of cross-section sampled data, whilst Spline methods can have overestimated local results.
Of higher importance is that the morphology of river channel beds is *anisotropic*, since the bathymetric variability is greater traverse to the flow than along it. Therefore, considering a flow-oriented coordinate system (Fig. 2) is a necessity to pursue a finer interpolated result. This allows for an anisotropy ratio, effectively bringing closer the points on the longitudinal direction. Towards this goal, in (Merwade et al., 2006) such a system is assumed and by evaluating different interpolation methods, it is shown to outperform the Cartesian coordinate system. For these reasons, spatial interpolation methods like EIDW have a higher interest to the covered topic.
However, the consideration of scarce input data is rarely accounted for when applying the aforementioned methods. In (Abebe, 2011, Osting, 2004), some insight is given on procedures that may be applied, but not enough information is conveyed on how to couple successfully an interpolation method to the physics of river morphology.

Methodology
In order to eventually assess the proposed method, datasets of full multibeam and relatively dense singlebeam river bed coverage were chosen. In order to correctly represent them in a structured way, a curvilinear grid is constructed from the river polygon's centreline. The centreline is acquired after cleaning the skeleton retrieved by the Voronoi tessellation of the river polygon's points. Consequently, the grid is constructed by extruding perpendicular lines to form the cells and the raw dataset is fitted on the grid cell edge points. This forms the ground truth dataset. From that, data points of cross-sections or tracklines can be automatically extracted to form the testing datasets (Fig. 3). Considering that the hydrographical parameters are available, calculated or an educated guess can be made about them, the physical model is computing the bathymetry based on Eq. (1) and Eq. (2). However, it is proposed to include bankline profiles based on the sampled data points close to the river polygon's edges (shallower parts).

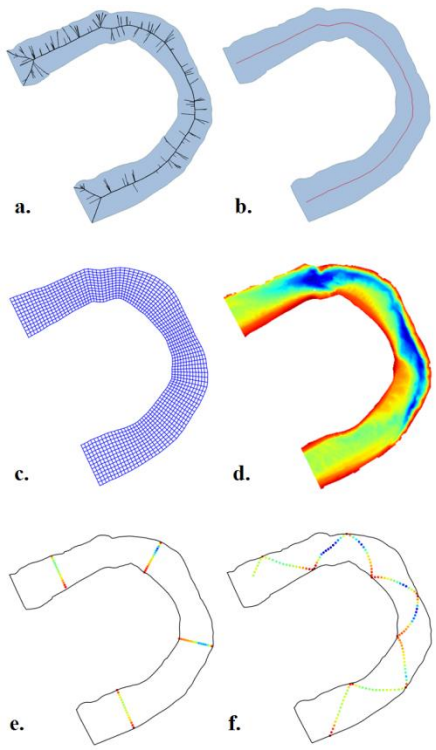

Figure 3. Grid construction and ground truth dataset: (a)Voronoi skeleton, (b)Centerline, (c)Curvilinear grid, (d)Ground truth dataset, (e)Cross-sectional testing dataset, (f)Trackline testing dataset.[Kootenai river, US].

In order to account for the natural process of advection and diffusion/dispersion of the river bed and to acquire a more realistic and continuous result, smoothing is applied to the model prediction using a gaussian window.
The spatial interpolation methods that suit the case of scarcely sampled datasets are Linear Baricentric Interpolation, Nearest Neighbour, IDW and EIDW. Their result alone however does not lead to acceptable depiction of the river bed.
To make the best out of the two results, a hypothesis by confidence of proximity is considered. Unsampled data that lie close to sampled data should consider the interpolated result more than the model's bathymetry and vice versa. This is modelled as weights in the range of [0,1] applied from each calculated dataset to each unsampled data point:

$$F_{(s,n)} = w_{(s,n)} B_{(s,n)} + (1 - w_{(s,n)}) I_{(s,n)} \qquad (3)$$

Where:

- $F_{(s,n)}$ – fusion result at (s,n)
- $B_{(s,n)}$ – model bathymetry result at (s,n)
- $I_{(s,n)}$ – interpolation result at (s,n)
- $w_{(s,n)}$ – model bathymetry weight at (s,n)

When the sampled data are in clear cross-sections the model bathymetry weight can be perceived as a linear gradient change from 0 on the cross-section to 1 in the middle between two cross-sections. For the case of trackline samples, a threshold distance value can be set, above which the weight for the model's bathymetry is considered 1.
The above comprises the "Fusion" method proposed, which for the purpose of this research merged the resulting datasets of EIDW and the model's bathymetry. All of the processes and computations were performed in the Python programming language.

Results

The method was applied to six river bends, two in each of the following rivers: Kootenai (US), IJssel (Netherlands), Danube (Romania). The testing datasets were iteratively thinned down to acquire a relation between amount of data and the method's performance in comparison to the spatial interpolation methods considered.
For Kootenai river (Fig. 4), a mostly natural river, the method performed best in terms of RMSE when available datasets of 7 or less cross-sections were available (Fig. 5), which corresponds to almost 200 m or larger of cross-section distances. For trackline data, a similar display was noted, with a threshold distance of 400 m.
The IJssel showed a similar tendency with larger distances between cross-sections (almost 400 m), but smaller thresholds for tracklines because of its smaller width.

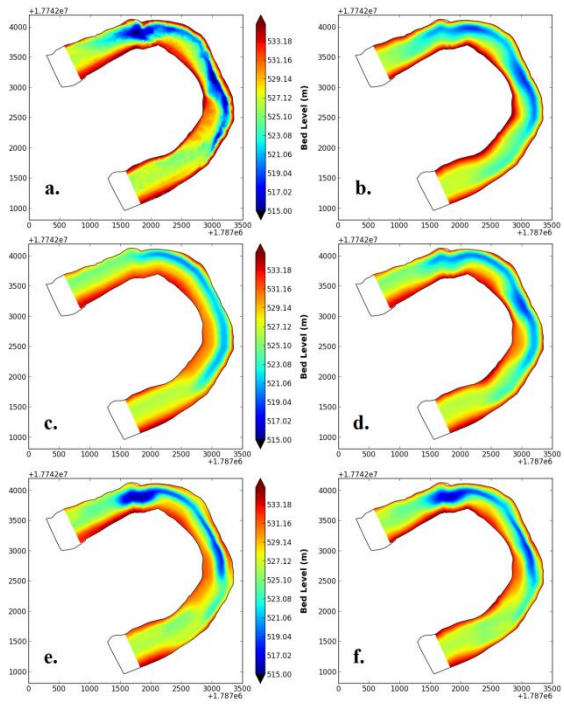

Figure 4. Example results of model, EIDW and Fusion method: (a)Ground truth, (b)Computed model bathymetry, (c)EIDW of Fig. 3e, (d)Fusion method of Fig. 3e, (e)EIDW of Fig. 3f, (f)Fusion method of Fig. 3f [Kootenai river, US].

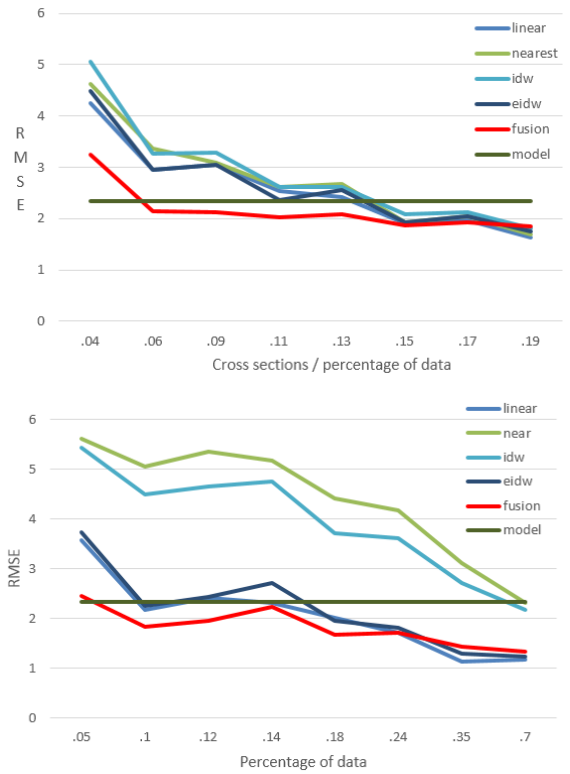

Figure 5. Example RMSE results for spatial interpolation methods, model bathymetry and fusion method. Top: Cross-sectional testing data, Bottom: Trackline testing data [Kootenai river, US].

However, in one of the two bends linear interpolation outperformed the rest, but with small differences to the RMSE values. That is due to the fact that it is a river where human intervention has been constant and various actions have been taken to shape its topography (groynes, dredging etc).

Finally, for the Danube river due to the limited dataset not a full evaluation was possible, but the general results showed a definite tendency towards a similar resulting RMSE graph.

In general, the EIDW and Linear methods were alternating as good candidates for coupling with the bathymetry model, however, the linear results tended to be too smoothened out.

To this extent, error maps like Fig. 6 were able to convey visually the areas of problematic predictions (overestimation/ underestimation), but also note where the method resulted in a less error-loaded general outcome.

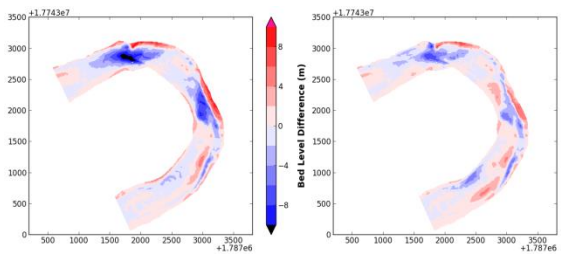

Figure 6. Error Maps of differences to ground truth of Fig. 4c,d. Left: EIDW, Right: Fusion method.

Conclusions and future work

The data samples dictate the result of an interpolation method, but coupled with a physical model a better and more natural outcome can be achieved. Of high importance are the transformation of the collected data to the flow-coordinate system and the consideration of the anisotropic nature of rivers.

For future work it is advised to expand the physical model with further features to involve extended cases (bifurcations, big variations in river widths), however staying within the prospect of a rapid assessment evaluation.

Acknowledgements

Special thanks go to Koen Berends, Ymkje Huismans and Erik Mosselman of Deltares (NL) for their guidance and skilful insight.

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Scour hole development Rhine-Meuse Delta (1967 – 2012) and future perspective

Y.Huismans, G. van Velzen, T. O'Mahoney, G. Hoffmans
Deltares, Boussinesqweg 1 2629 HV Delft, the Netherlands

Introduction

The Rhine-Meuse estuary is the area where the Rhine and Meuse discharge into sea. Located in the most densely populated and intensively used part of the Netherlands, bank stability, sufficient coverage on tunnels and cables, and enough navigable depth are of high importance. This requires proper managing of the riverbed and a profound knowledge on the riverbed dynamics, which in this area is largely governed by its subsoil composition and anthropogenic influences.

The subsurface lithology in this tidal area is very heterogeneous (Berendsen & Stouthamer, 2001; Cohen, Stouthamer, Pierik, & Geurts, 2012; Hijma, 2009). Layers of poorly erodible peat and clay alternate with highly erodible sand bodies, as illustrated in Figure . When thinner parts of the clay or peat layers of the river bed erode, the underlying sand patches are incised and large scour holes can develop (Sloff, van Spijk, Stouthamer, & Sieben, 2013). These scour holes form a potential risk for the stability of nearby structures like dikes and groynes. This process is believed to have strengthened due to the closing of the Haringvliet Sluices (1970), which increased the flow velocities in the connecting branches in the central part of the network [(Sloff et al., 2013; Vellinga, Stouthamer, Wang, Sloff, & Hoitink, 2015)].

In this research ((Ymkje Huismans, O'Mahoney, van Velzen, & Hoffmans, 2015)) we analyse single-beam and multi-beam measurements from 1967 up to present to investigate the historic evolution of the scours in the Rhine Meuse Delta. With this an overview of the initiation and further growth of scour holes has been acquired, which is subsequently linked to various anthropogenic influences and the subsurface lithology. For a selection of scour holes a detailed analysis has been performed, with an outlook on its future development based on the recent scour evolution, theoretical formulations for scour evolution and detailed information on the subsurface lithology. Finally results are compared to recent insights from scale model tests on scouring in heterogeneous subsoil (Zuylen & Sloff, 2015).

Data

For this research bed topography data from 1967 up to 2012 were analysed. Before 2005 this data are based on single-beam echolood measurements, from 2005 they are based on multi-beam echolood measurements.

Knowledge on the subsurface lithology was taken from two datasets: a GIS-map with the location and dates of the channel belts (Cohen et al., 2012) and detailed maps of the subsurface lithology under the river bed (Sloff, van den Ham, Stouthamer, & van Zetten, 2011; Wiersma, 2015).

To get an estimation of the flow velocities nearby the scour holes, depth

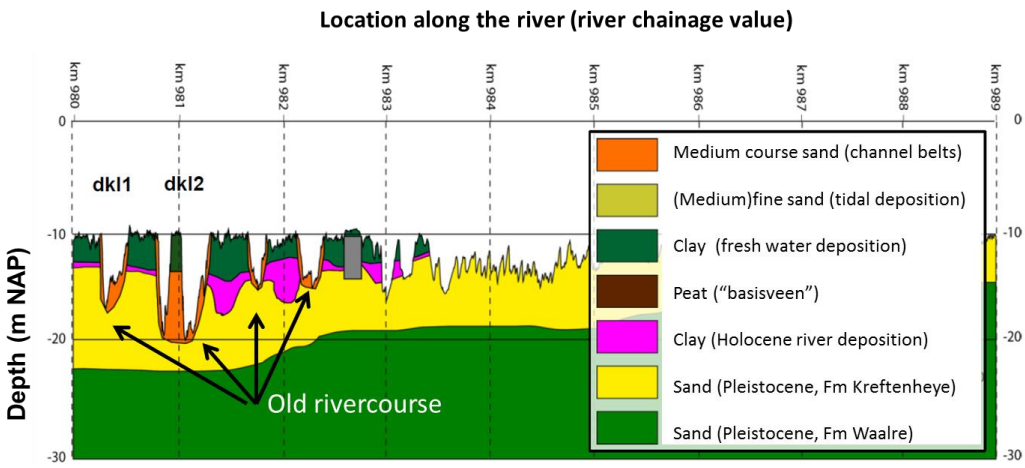

Figure 1 Subsurface lithology below the Dordtsche Kil river. The current bed topography is imprinted on this longsection. The different layers have formed during different geological eras. Old river courses (in orange) have deposited sand. These rivers may have crossed the current river course as is the case in the Dordtsche Kil. Remaining channel belts show up as patches of sand in the clay and peat layers. Note that the location of all layers and channel belts are based on bore holes, river bathymetry and geological interpretations, as such there is a high uncertainty in their exact location.

averaged flow velocities from the Rhine-Meuse quasi 2D morphodynamic model where taken, for low, medium and high river-discharges (Ottevanger & van der Mark, 2015).

Scour holes from 1967 to present

Method

Based on the 2012 multi-beam data, scour holes in the area were identified, which amounts to nearly 100 scour holes. For the most important branches[1] the scour holes (30) have been divided into three categories:

A. Length smaller than 200 m, not located nearby a structure (often younger scour holes).
B. Length larger than 200 m, not located nearby a structure (often older scour holes).
C. Located nearby a structure (any length).

With the bed topography data from 1967 to 2012, their historic evolution has been mapped.

Results

From the classification it follows that the majority of the scour holes cannot be related to the presence of a structure (67%), see Figure . This is remarkable as the Rhine Meuse Delta is a heavily engineered area, which contains many structures. This shows the relative importance of the other drivers for the formation of scour holes, like the subsurface lithology.

The distinction between scour type A and B was made with the expectation that smaller scour holes are often the younger scour holes and the larger scour holes the older ones. The analysis however shows that most of the small scour holes are decades old. A possible explanation is found in the composition of the local subsurface lithology; when these scour holes are located in an old channel belt they are constrained in growth by the presence of thick layers of poorly erodible clay or peat at the edges (see Figure). If this channel belt is small, the resulting scour hole may stay small as well.

Previous research has shown that the flow velocities in the connecting branches have increased as a result of the closure of the Haringvliet ((Sloff et al., 2013; Vellinga et al., 2015)). This has led to the assumption that the closure of the Haringvliet has been an important driver for the genesis of the scour holes. From Figure it can however be concluded that most scour holes (60%) were already present before the closure of the Haringvliet. This does not mean that closing the Haringvliet had no effect on the further development of the scour holes. More research will be carried out to verify whether the development in depth and extent has been accelerated in the years after closure.

All Dordtsche Kil scour holes have emerged within a remarkably short time, between 1976 and 1979. This is not only the period just after the closure of the Haringvliet, it is also the period during which the Dordtsche Kil was reconstructed. Based on the short timespan in which the scour holes emerged, it is most likely that reconstruction works have initiated the scour holes rather than closure of the Haringvliet. By dredging the river to -8 m NAP (nowadays even -10 m NAP) the protecting clay layer was most likely removed uncovering the sand bodies from old channel belts, see Figure .

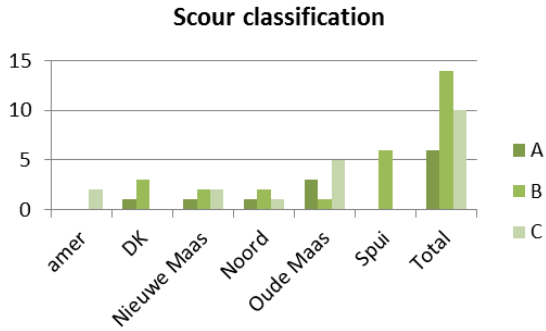

Figure 2 Scour hole classification into categories A, B and C (see method-section).

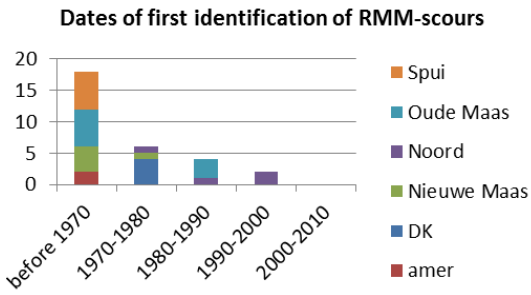

Figure 3 Dates of the first identification of scour holes in the single- or multi-beam echolood measurements.

Detailed analysis on scour evolution

Method

To get a better understanding on the scour hole evolution and their potential future development, a detailed analysis has been performed for a set of six scour holes: two Dordtsche Kil scour holes, one in the Noord, two in the Oude Maas and one in the Spui. The following steps were followed:

1) Analysis of the general scour hole development from 1970 to 2012.
2) Detailed analysis of the growth in depth, extend and the evolution of the slopes over the last 7 years.

[1] *In consultation with the Dutch Water Authority (RWS) the following branches are regarded most important: Oude Maas, Nieuwe Maas, Noord, Dordtsche Kil, Spui and Amer.*

3) Outlook on scour hole evolution based on the recent scour hole evolution, the subsurface lithology and the expected scour hole depth as calculated with the expression for equilibrium depth (Hoffmans, 2012).

Results

In this paper, one scour hole will be discussed in more detail, so as to illustrate the method. A more global discussion will follow for the other scour holes.

The Dordtsche Kil contains four identified scour holes. The second one ("dkl2", counted from north to south), consists of two scour holes separated by a presumably poorly erodible clay layer, see Figure for its bed topography and Figure for the surrounding geology. The scour hole has presumably formed in an old channel belt, when the poorly erodible top layer of the Dordtsche Kil was removed when deepening the navigation channel. Note that the locations of the channel belts are optimized based on the morphology of the river bed, local measurements are needed to verify that this scour hole is indeed located in a channel belt.

The northern part of the scour hole originated somewhat after 1976 and showed a gradual growth in depth and extend. Even over the last few years a steady growth is observed and the clay layer between the northern and southern scour hole seems to be depleted slowly. The expectation for the future is that this scour hole will grow and exceed its current depth of -20.5 m NAP. Based on calculations, its equilibrium depth would be even -27 m NAP. These calculations are however based on limited information on hydrodynamics and critical shear stresses. They furthermore do not regard changes in bed material. At a depth of about -23 m NAP a clay layer is located (the exact location is uncertain as it is based on an interpolation of bore holes). The growth in depth will most likely be stopped by the top of this layer. The layer separating the two scour holes may get undermined in the future. It should finally be noted that slopes are very steep, exceeding values of 1:1, this poses potential risks for undermining.

From the other analysed scour holes we can conclude the following:

- Scour holes may merge and form a large trench, attracting flow and possibly further accelerate scour hole development.
- The subsurface lithology is most likely the most important predictor for future scour hole evolution.
- Slopes are often exceeding 1:1, this poses potential risks for undermining.
- Most scour holes have been initiated decades ago and are either stable or show a gradual evolution in depth and extend. Only few scour holes in the area have recently emerged.

Initial development new scour holes and relation to scale model tests

The initial development of new scour holes is of much interest, as their speed of development determines the time for intervention and monitoring. In the Oude Maas a new scour hole has formed in 2005 and shown a fast development in the subsequent years of up to tens of meters per year in length and a depth development from no scour hole (-17 m NAP) to a depth of -27 m NAP in 2013, see Figure , (Y. Huismans, van der Wal, & Beekhuizen, 2015). This shows that the initial development goes very fast and that the

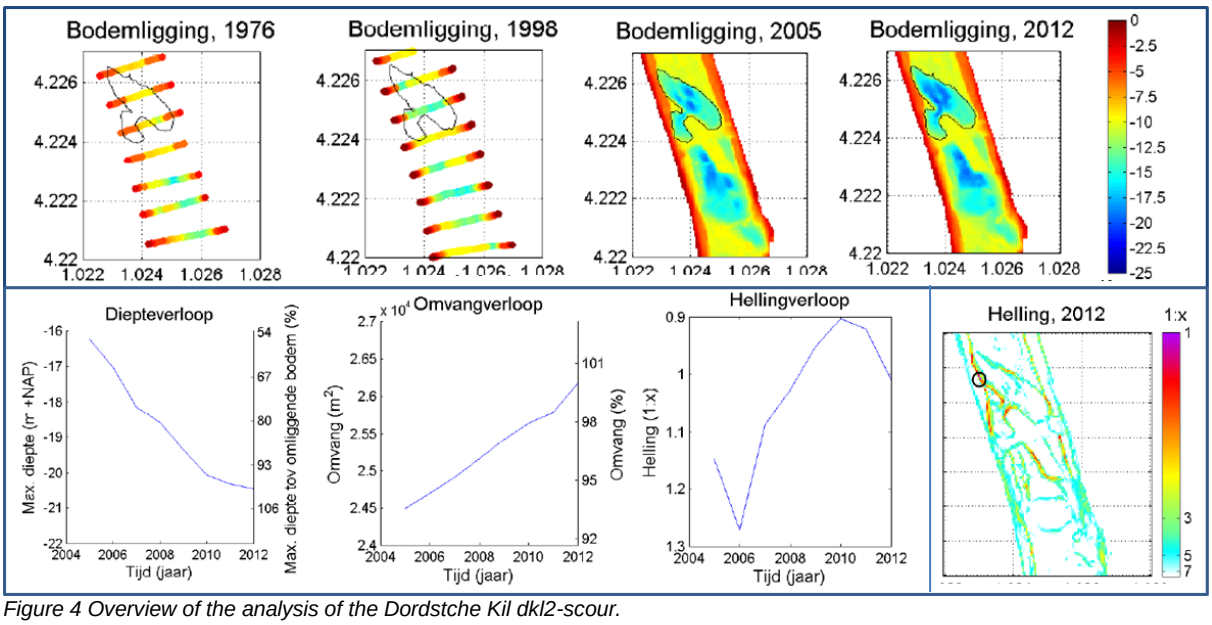

Figure 4 Overview of the analysis of the Dordstche Kil dkl2-scour.

fastest expansion and steepest edge is in the predominantly upstream direction (the Oude Maas is a tidal river). This is in contradiction with results from recent scale model tests, which showed the opposite (Zuylen & Sloff, 2015). More research will be carried out to clarify these results.

Conclusions

Bed topography data from 1967 to 2012 have been analysed to get an overview of the scour hole evolution in the Rhine Meuse Delta. Results have been related to the subsurface lithology and human interventions like the closure of the Haringvliet and dredging activities. From this analysis it can be concluded that most scour holes were already present before closing the Haringvliet. This nuances the hypothesis that the related changes in hydrodynamics induced most scour holes. This does however not exclude that existing scour holes did undergo acceleration in development due to the closing of the Haringvliet.

According to the analysis it is most likely that the scour holes in the Dordtsche Kil originated as a result of reconstruction works. By removing or thinning the poorly erodible top layer by dredging works, underlying sand patches have been incised, leading to large scouring within in few years timespan. This stresses the importance of having firm knowledge on the subsurface lithology before deepening a channel or digging a new one.

Most scour holes are seemingly stable; however gradual evolution may lead to significant changes in the long term, to even merging of scour holes to a trench. This may subsequently lead to an acceleration of the scour hole evolution and to undesirable flow velocities for navigation.

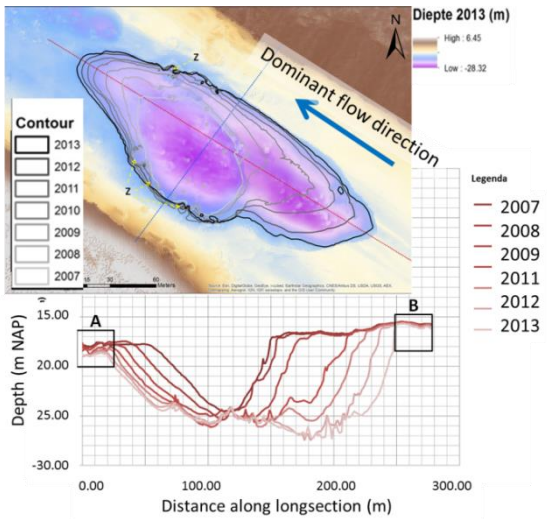

Figure 5 Development of a scour hole in the Oude Maas, close to the Beerenplaat. Top figure shows the local bed topography. The contours show the scour area in the subsequent years. The bottom figure shows a longsection of the bed topography.

Close monitoring and stabilizing scour holes is therefore of high importance.

The evolution of a recently formed scour hole shows a rapid expansion in depth and length. The fastest expansion is in the upstream direction. This is in contradiction to recent scale model tests. More research will be carried out to clarify these results.

Outlook

Currently all scour holes in the area (~100) are analysed on their recent development in depth, area and slope. This systematic analysis is expected to enhance our understanding of the time-evolution of scour holes that are located in a heterogeneous subsurface.

Acknowledgements

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Consolidation and strength development by horizontal drainage of soft mud deposits in lake Markermeer

Maria Barciela Rial*[1], Johan C. Winterwerp [1,2], Jasper Griffioen[3,4] and Thijs van Kessel[2]
[1] *Delft University of Technology, Faculty of Civil Engineering and Geosciences, Stevinweg 1, 2628 CN Delft, The Netherlands*
[2] *Deltares, PO Box 177, 2600 MH Delft, The Netherlands*
[3] *Department of Innovation, Environmental and Energy Sciences, Faculty of Geosciences, Utrecht University, P.O. Box 80115, 3508 TC Utrecht, The Netherlands*
[4] *TNO, P.O. Box 80015, 3508 TA Utrecht, The Netherlands*
* Corresponding author; e-mail*: m.barcielarial@tudelft.nl*

Introduction

The behaviour of soft, muddy sediments is becoming increasingly important, as large amounts of mobilised sediments will progressively be used for nature building projects. These fine sediments represent a greater challenge than traditional sandy ones, because of their diverse properties.

In the Netherlands, various building with nature (BwN) projects have already been implemented (De Vriend et al., 2015) with different purposes: from coastal safety (Zand Motor, The Hague) to protection of eroding intertidal shoals with oyster reefs in the Eastern Scheldt (Zeeland).

The MarkerWadden is an example of an ongoing BwN project which aims to improve the ecosystem in lake Markermeer (The Netherlands) by creating islands, marshes and mud flats with sediments partly originating from the fluffy material of the bed of the lake itself. It represents one of the first projects which use fresh unconsolidated mud as a construction material.

Figure 1. Present situation of Markermeer (www.wikipedia.com). The new polders and dikes (from the Zuiderzee works) can be observed in the figure.

Lake Markermeer (Fig.1) is large and shallow: its surface is 680 km^2 (including the IJmeer and the Gouwzee) and its average water depth is 3.6 m (Rozari, 2009; Vijverberg et al., 2011). A thin fluffy layer of silt dominates the lake bed. Already at low wind speeds, wind-induced-waves cause resuspension of this top layer. As a result, the high concentration of suspended particles inhibits light penetration causing the deterioration of the surface water quality. Below this thin fluff layer, a thicker layer of fluvial mud which has been deposited after closure of the Afsluitdijk is observed. Below this layer, a base of marine deposits is present, originating from the period before closure.

Problem analysis

Figure 2 presents a schematic diagram of wetland building with soft mud showing that part of the soil experiences "classical" consolidation with vertical drainage due to self-weight consolidation (column I, Fig. 2). However, higher on the wetland, the soil (water-sediment mixture) rises above the water level, and pore water is more likely to escape also in horizontal direction (column II, Fig. 2) because of the local slope at the water table.

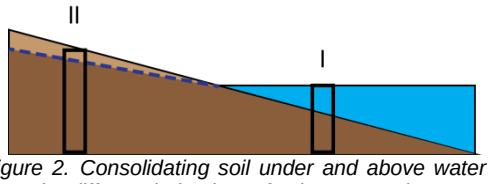

Figure 2. Consolidating soil under and above water – Note the different behaviour of columns I and II

Experimental Methods

A variety of experiments is applied to determine the consolidation and strength development of fresh mud deposits from lake Markermeer. This novel approach mimics land and crust formation with soft soils. The influence of the crust and sediment variability on the consolidation process is studied, as well as the physical-chemical properties. Afterwards, a model is used to upscale the results. This method provides engineering rules for wetland creation and contributes to the understanding of the dominant mechanisms for soil formation from soft sediments.

Previous to the beginning of the horizontal drainage experiments, three small settling columns (each with a volume of 2 litres) with three different concentrations (below the gelling point) are used to determine the sediment properties (i.e., bulk permeability k as a function of void ratio e and void ratio e as a function of and vertical effective stress σ_{zz}^{sk}) by monitoring the settlement of the sediment interface in time. These concentrations must be below the gelling concentration, which represents the concentration at which flocs become space-filling and form a network structure or gel and measurable shear strength builds up (Dankers, 2006).

Once the properties of the soil are known, three types of columns are needed (see fig. 3) in order to perform a horizontal drainage experiment which is properly calibrated. The first one is a control column without any drainage system. In this column, only vertical drainage due to overburden occurs. It gives us the properties of the material. A second column, equipped with a Vyon porous pipe, gives us the effect of the pipe without drainage. Finally, a third column, also equipped with an identical porous pipe, which is now connected to a reference water table by a hose, allows us quantify and observe the effects of horizontal drainage, i.e. an extra difference in head. The length of the porous section of the pipe may be changed from zero to the full column height. In this way, the section of the column experiencing horizontal drainage may be varied.

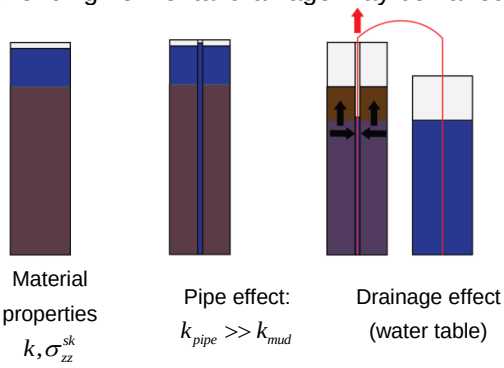

Figure 3.Calibration of the drainage of the columns

Mathematical modelling

An equation for the consolidation of a sloping bed is derived for upscaling the results of the experiments. Note that, for the time being, the equation presented in this abstract does not account for chemico-biological effects and precipitation/ evaporation. However, the influence of these parameters can be experimentally calibrated and included in the formulation.

For the derivation of this equation, the Eulerian approach by Merckelbach and Kranenburg (2009) is followed, (see also Winterwerp and Van Kesteren, 2004). Figure 4 shows a sketch of a consolidating column of fine sediment, and the various velocities of the particles and pore water. Horizontal drainage is indicated by w_f. It is hypothesized that w_f may change over time and with depth due to consolidation, but remains constant in lateral direction y. Hence, it is assumed that w_f and the permeability k are a function of the vertical coordinate z and time t only. The parameters involved in Fig. 4, as well as the stresses playing a role in the process, are defined in Table 1.

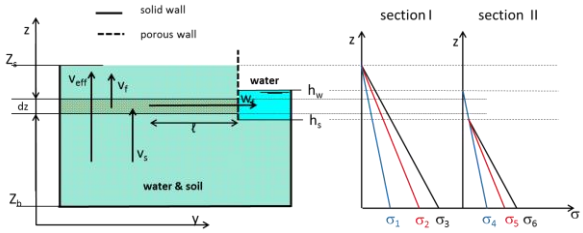

Figure 4. Schematization of consolidating soil and vertical stress distribution

Table 1. Definition of parameters and stresses of Figure 4.

v_f	vertical fluid velocity in Eulerian frame	
v_s	vertical settling velocity of solid particles	
v_{eff}	effective fluid velocity through porous soil	
w_f	horizontal drainage velocity	
ℓ	characteristic length scale for horizontal drainage	
Z_s	level soil-water mixture	
Z_b	bottom level	
h_w	water level	
h_s	level permeable wall	
ρ_b	bulk density	
σ_1	hydrostatic pressure	$\sigma_1 = g\rho_w(Z_s - z)$
σ_2	actual pore water pressure	p_w
$\sigma_2 - \sigma_1$	excess pore pressure	p_e
σ_3	total stress	$\sigma_{zz} = g\int_z^{Z_s} \rho_b \, dz$
$\sigma_3 - \sigma_2$	effective stress	σ_{zz}^{sk}
σ_4	hydrostatic pressure	$\sigma_4 = g\rho_w(h_w - z)$
σ_5	actual pore water pressure	p_w
σ_6	total stress	$\sigma_{zz} = g\int_z^{h_w} \rho_b \, dz$
$\sigma_5 - \sigma_4$	excess pore pressure	p_e
$\sigma_6 - \sigma_5$	effective stress	σ_{zz}^{sk}

Horizontal drainage takes place above level h_s, and it is driven by the head difference between Z_s and h_s.

Elaborating on the 2D continuity equation for the solid fraction ϕ, the momentum equations in y- and z-direction and the vertical gradient stresses, we obtain the consolidation equation (1) for fine sediments with horizontal drainage:

$$\frac{\partial \phi}{\partial t} - \frac{(\rho_s - \rho_w)}{\rho_w}\frac{\partial k\phi^2}{\partial z} - \frac{\partial}{\partial z}\left(\frac{k\phi}{g\rho_w}\frac{\partial \sigma_{zz}^{sk}}{\partial z}\right) = \\ = \frac{\mathrm{d}Z_s}{\mathrm{d}y}\frac{\partial k\phi}{\partial z} \qquad (1)$$

where ρ_s is the specific density of the soil.

The left-hand side of equation (1) represents the classical one-dimensional equation for self-weight consolidation and can be used to determine the material properties. The right-hand side defines the effect of the bed slope.

Next, we introduce the fractal descriptions for the mud permeability and effective stress (e.g. Merckelbach and Kranenburg, 2004; Winterwerp and Van Kesteren, 2004):

$$k = K_k \phi^{-2/(3-n_f)}$$
$$\sigma_{zz}^{sk} = K_p \phi^{2/(3-n_f)} \qquad (2)$$
$$\Gamma_c = \frac{2}{3-n_f}\frac{K_k K_p}{g\rho_w}$$

in which n_f = fractal dimension, K_k [m/s] and K_p [Pa] are coefficients for permeability and effective stress. Γ_c represents a consolidation coefficient, which equals the classical coefficient c_v. Substitution of equation (2) into (1) yields an advection-diffusion equation (3):

$$\frac{\partial \phi}{\partial t} - \Delta_\rho \frac{\partial k\phi^2}{\partial z} - \Gamma_c \frac{\partial^2 \phi}{\partial z^2} = \frac{\mathrm{d}Z_s}{\mathrm{d}y}\frac{\partial k\phi}{\partial z} \qquad (3)$$

where $\Delta_\rho = (\rho_s - \rho_w)/\rho_w$

Note that, in order to apply these equations to the explained experimental setup (cylindrical columns), all of them are first rewritten in cylindrical coordinates.

Conclusions

With the described experimental and mathematical methods, the characteristic consolidation parameters of the clayey soil can be obtained. Moreover, the effects of vegetation, evaporation/precipitation and organic geochemistry can be included in the equation (or boundary conditions or material parameters) in the future. Finally, the derived equations for consolidation with horizontal drainage can be implemented in a 2D transport model, such as Delft 3D (Zou et al., 2015).

Thus, this method represents a powerful tool which can be used to develop engineering rules for wetland creation from soft sediment.

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Tidal motion and salt dispersion at a channel junction

M. C. Verbeek*1, A.J.F. Hoitink1 , B. Vermeulen1
[1] *Wageningen University, Hydrology and Quantitative Water Management Group, Department of Environmental Sciences, Building 100, Droevendaalsesteeg 3, 6708PB WAGENINGEN, the Netherlands*
** Corresponding author; e-mail: verbeekmc@gmail.com*

Introduction

On a global scale, freshwater availability in low-lying coastal areas is threatened by salt water intrusion. There is a need to better understand the processes governing salt intrusion, especially in densely populated deltas. In delta channel networks, channel junctions may play a key role in salt dispersal.

This study analyses the tidal dynamics and salt dispersion at a channel junction in the Rijn-Maas Delta in the Netherlands, using a 13-h survey of Acoustic Doppler Current Profiler (ADCP) and Conductivity Temperature Depth (CTD) data. The study includes a unique set of five salinity profiles in each of the three cross-sections bounding the Hartelkanaal - Oude Maas junction, once every hour (Fig. 1) .

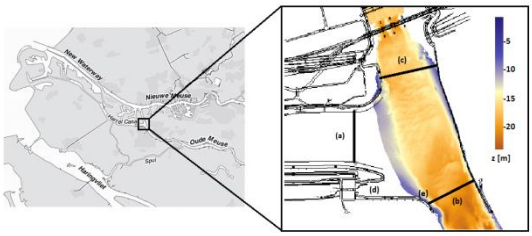

Figure 1. (left) The Rijn-Maas delta, (right) The studied channel junction, the black lines are the tracks of the boat. From south to north: Oude Maas, west: Hartelkanaal.

The observations capture the 3D character of the flow and salt fluxes at the junction. This study aims to provide more insight in the intra-tidal processes, which disperse salt at tidal junctions.

Tidal fit

A velocity variation model (tidal model) of the semi-diurnal and fourth-diurnal tidal wave was successfully fitted through the raw velocity data, which made it possible to determine the tidal amplitude and tidal phase for every position at the cross-sections. This elaborates on the method proposed by Vermeulen *et al.* (2014).
The tidal model was defined as:

$$u = u_0 \cdot 1 + A_{M_2} sin(f_{M2}t) + B_{M_2} cos(f_{M2}t) + A_{M_4} sin(f_{M4}t) + B_{M_4} cos(f_{M4}t), \quad (1)$$

where u is the velocity, f_{M2} and f_{M4} are the imposed astronomical frequencies for the semi-diurnal and fourth-diurnal tidal wave, A and B result from the fit and are used to calculate tidal amplitude and phase. u_0 is the residual velocity.

In figure 2. an example fit is given for the velocity time series of one cell in the Oude Maas transect.

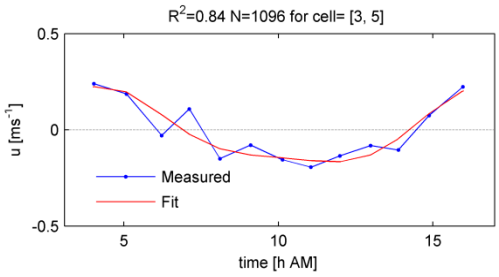

Figure 2. The velocity time series for one cell in the Oude Maas with an average R^2. The blue dots are the velocity data points, the red line is the fitted tidal model.

The flow at the junction is dominated by the semi-diurnal tide, which shows to be highly spatially variable (Fig. 3). The tidal phase differs 1 to 2 hours between the junction branches, and locally phase lags of up to 4 hours are found.

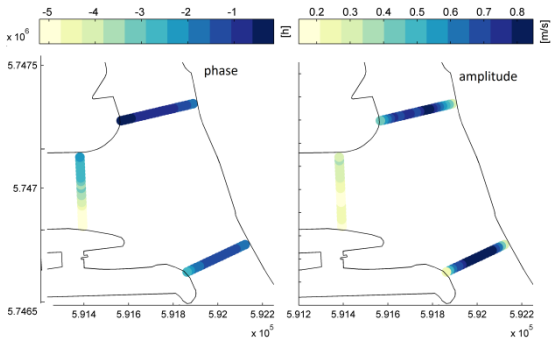

Figure 3. (left) The tidal ellipses of the dominant M_2 tide in m/s, (right) The depth averaged tidal phase of the M_2 tide in hours.

Decomposition salt transport

The residual salt transport was decomposed into contributions representing (1) the advective transport due to water discharge and change in storage during the tidal cycle, (2) a sloshing effect, i.e. tidal dispersion via triple correlation between tidal depth change, tidal current, and tidal salinity, (3) the cross-correlation between tide and salinity, (4) Stokes' drift dispersion and (5) salt dispersion due to mean shear produced by gravitational circulation.

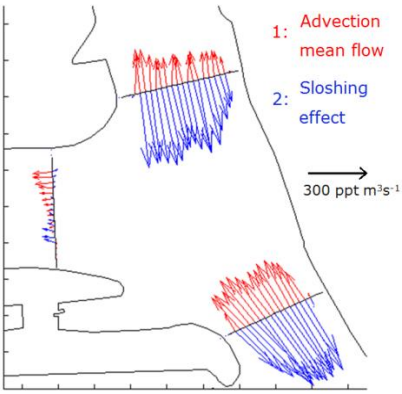

Figure 4. The two main contributors of the salt transport by 1) advection and 2) sloshing over the studied channel junction.

Advective transport and the sloshing effect dominate the salt balance (Fig. 4). This implies that salt intrusion is largely controlled by the character of the barotropic tidal motion.

This study shows that the model of a salt balance dominated by estuarine circulation is not always applicable to multi-channel estuaries where the salt dispersion has a more 3D character than in single-head estuaries.

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Establishing a sediment budget in the 'Room for the River' area 'Kleine Noordwaard'

E.C. van der Deijl*[1], E. Verschelling[1,2], M. van der Perk[1], H. Middelkoop[1]

[1] *Utrecht University, Department of Physical Geography, Faculty of Geosciences, PO Box 80115, 3508 TC Utrecht, The Netherlands*

[2] *Deltares, PO Box 177, 2600 MH Delft, the Netherlands*

** Corresponding author; e-mail: E.C.vanderDeijl@uu.nl*

Introduction

Many deltas in the world cope with drowning and loss of delta land by sediment starvation and accelerated soil subsidence, because of embankment of channels and drainage of land. The urgency of the problem is enhanced by sea level rise (Syvitski et al., 2009). Loss of delta land is a problem, since most deltas are densely populated and seen as valuable because of their ideal location for harbours, agriculture, aquaculture, or tourism (Kirwan and Megonial, 2013). Moreover, deltas encompass vast wetland areas of great ecological value. Delta restoration by re-introduction of natural processes and sedimentation is considered as a mitigation measure, but it is difficult to implement, since natural processes are often in conflict with current activities in the delta. Furthermore, several deltas are subject to sediment starvation, because of a decrease in sediment delivery from upstream. For these delta's it is uncertain whether delta restoration will be effective.

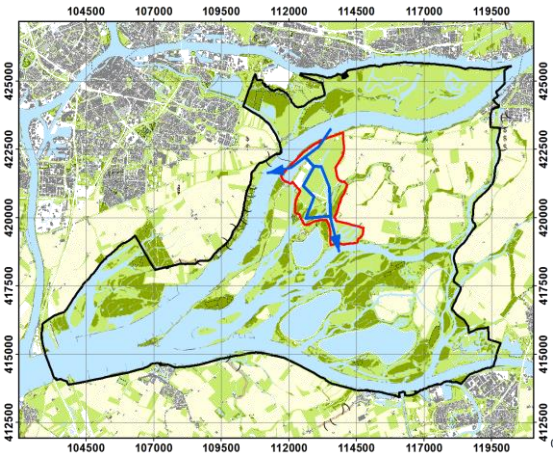

Fig.1. The Biesbosch (black) and the study area Kleine Noordwaard (red) with its main flow pathways.

Effective delta restoration requires a thorough understanding of the mechanisms of delta aggradation and their controls. In the Biesbosch (Fig.1), an inland delta in the south-west of the Netherlands, water and sediment is reintroduced in former polder areas as part of a large project for improving the discharge capacity of the lower Rhine branches ('Room for the River' (RfR)). This makes the Biesbosch the ideal trial area to study the mechanisms and controls of delta aggradation.

Aim and Method

This study aims to quantify the amounts and spatial patterns of aggradation in the former polder area 'Kleine Noordwaard' (Fig.2), in which water and sediment have been reintroduced since 2008. The following data were used:

4. Channel bathymetry using Multibeam echo sounder data
5. Thickness of mud or sand deposited on the compact, former polder soil surface, sampled by a transparent corer tube.
6. Location and height of cut banks at the island measured using a dGPS device and ruler.
7. Digital elevation model (Actueel Hoogtebestand Nederland)

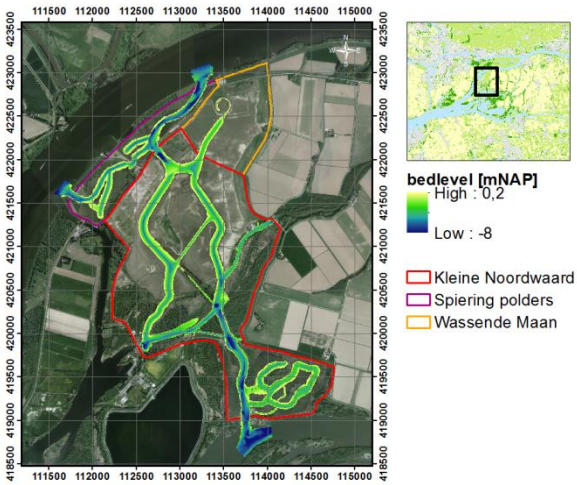

Fig.2. The bathymetry and subdivision of study area 1, Kleine Noordwaard.

Results

Channels

Consecutive measurements of channel bathymetry show a positive sediment budget in the channels. Fig.3 shows the difference in height between March 2009 and March 2012. During this period the total sediment budget of the channels in Kleine

Noordwaard was 59.5 10^3 m^3. Deposition in the channels was on average 19.8 10^3 m^3/year.

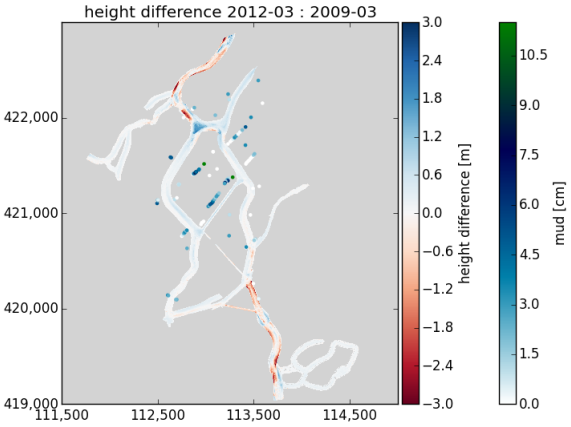

Fig.3. Difference in bed elevation between March 2009 and March 2012. Dots represent mud accumulation on the intertidal flats between May 2008 and October 2014.

Fig.4 shows the cumulative change in channel bed volume from north to south in the 'Kleine Noordwaard' for the successive monitoring campaigns. A decrease in channel bed volume was observed at the entrance and exit of the system. Most erosion in Spiering polders takes place between September 2010 and March 2011, triggered by the peak river discharge of 8315m^3/s at Lobith, between 8-19 January, 2011 (Rijkswaterstaat,2014).

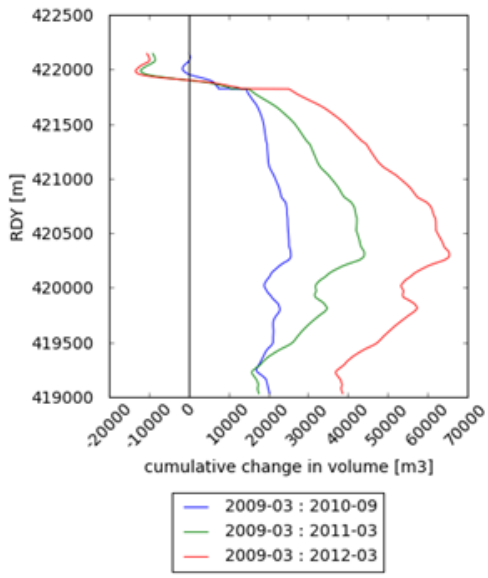

Fig.4. Cumulative channel bed volumes within the Kleine Noordwaard, along a N-S transect starting from the Spiering polders (purple in Fig.2) at y –coordinate 422366 to south. The budget of the Wassende Maan (orange in Fig.2) is added at y-coordinate 421821m.

Around y-coordinate 421940, the cumulative change in bed volume turns positive; suggesting that the vast majority of the sediment eroded in the Spiering polders and near the entrance of the Kleine Noordwaard was deposited within 420m from the entrance. Furthermore, a large increase in channel bed volume was observed in the central part of the area. The pattern of erosion and deposition across the polder was persistent over the years.

Intertidal flats

Approximately 21 mm of mud accumulated in the intertidal area between May 2008 and October 2014 (Fig.3). This corresponds to an accumulation rate of 3.2 mm/yr.

Sand is mainly deposited near the entrance of the system and close to the channels, while deposition of mud varies locally.

Cut banks

Comparison of the current cut bank position and height with the digital elevation model of the area indicates that only 31 m^3 sediment eroded from the island between 2009 and 2014, which is not significant compared to the sediment deposition and erosion volumes in the channels and on the intertidal flats.

Conclusion

The total sediment budget of the 'Kleine Noordwaard' area amounted to 27.4 x 10^3 m^3/yr which corresponds to a net area-average sedimentation rate of 4.7 mm/yr during the first 5 years after depoldering of the area. The channels received 19,1 x 10^3 m^3/yr sediment and the intertidal area 8.3 x10^3 m^3/yr. Remobilization of sediment by erosion of cut banks occurred at a negligible rate of about -6 m^3/yr.

Acknowledgements

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Longitudinal training walls: morphodynamic effects of the starting point

T.B. Le*[1, 2], A. Crosato[1, 3], W.S.J. Uijttewaal[1]
[1] *Delft University of Technology, PO Box 5048, 2600 GA Delft, the Netherlands*
[2] *Water Resources University, 175 Tay Son, Dong Da, Hanoi, Vietnam*
[3] *UNESCO-IHE, Department of Water Engineering, PO Box 3015, 2601 GA, Delft, the Netherlands*
** Corresponding author; e-mail: T.B.Le@tudelft.nl*

Introduction

The construction of a longitudinal training wall in a river creates a side channel parallel to the main channel (Figure 1) distributing water flow and sediment transport. The morphodynamic evolution of both main and side channels is affected by these distributions, which are in turn affected by the presence of bars (Hansen 1967; Engelund & Skovgaard 1973). In low-land rivers the most common bar type is alternate bars and due to the presence of permanent geometrical variations in the river channel, these bars are typically steady (Struiksma et al. 1985, Struiksma & Crosato 1989). The effects of longitudinal training walls were studied by Le et al. (2014) based on a series of numerical simulations. The results suggest that the starting point of the longitudinal training wall with respect to a bar might play an important role for the morphological developments of both main and side channels and this is true also for point bars inside river bends. However, there were no field data or laboratory test results available to validate this finding. So, it is now necessary to carry out an experimental study to test what Le et al. (2014) found.

The work presented here analyses the effects of a longitudinal training wall with different starting points in an experimental flume (Figure 2). The results of these experiments are still preliminary and the work is still going on.

Figure 1. Longitudinal training wall between Wamel and Ophemert. Source: Rijkswaterstaat.

Methodology

The experimental tests are carried out in the Fluid Mechanic Laboratory of Delft University of Technology. The reference case is a channel with steady alternate bars. Other test cases regard the same system with a training wall parallel to the glass side-wall of the flume and differ from the starting location of the training wall with respect to a steady bar.

The preliminary part of the experiments regards two locations only, one near the upstream and one near the top of a bar. This paper describes the results of the first tests carried out without sediment feeding.

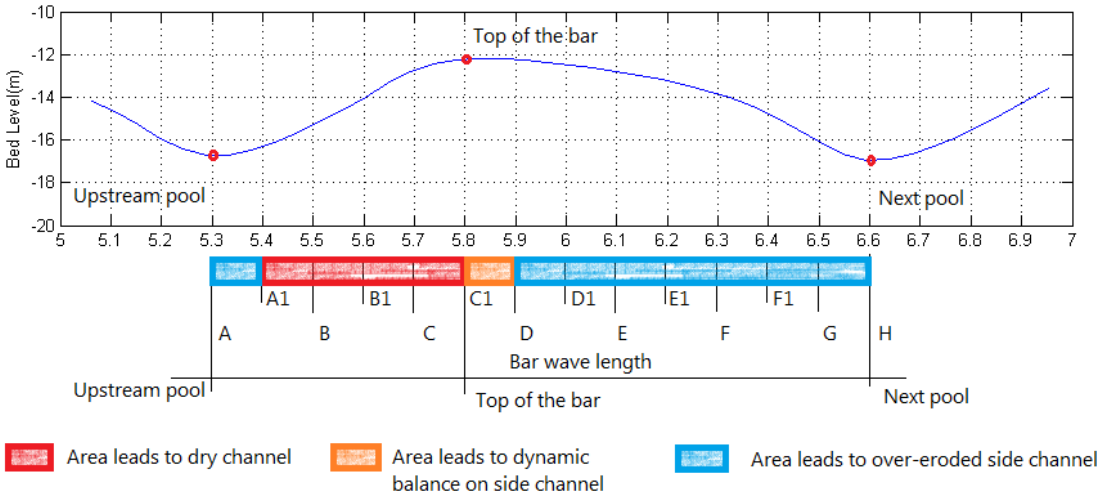

Figure 2. Effects of longitudinal training wall's starting point on side channel development (Le et al., 2014).

Experimental setup

Based on Crosato-Mosselman's (2009) formula (Equation 1), a straight channel with geometrical (width, depth, slope) and morphodynamic (flow and sediment) characteristics is selected having bar mode = 1 (alternate bars).

$$m^2 = 0{,}17g\frac{(b-3)}{\sqrt{\Delta D_{50}}}\frac{B^3 i}{CQ} \qquad (1)$$

Where: m is the bar mode, b is the degree of non-linearity of the sediment transport as a function of the flow velocity, B is the river width, i is bed slope, Δ is sediment relative density, D_{50} is sediment mean size, C is the Chézy coefficient and Q is river discharge.

The total length of the experimental flume is 14.4 m; the width is 40 cm. The channel bed is covered by a layer of sand having thickness of 20 cm. The mean diameter of the sediment, D_{50}, is equal to 0.37 mm. The water discharge is kept constant at a value of 4.0 l/s. A floating sponge is used to reduce free surface disturbances at upstream of the flume.

Steady bar formation is obtained by placing a transverse plate, 26 cm long, at the upstream cross-section of the channel, which obstructs 2/3 of the channel width (Figure 3). This creates a permanent forcing due to flow non-uniformity at the inflow leading to the formation of steady bars (Struiksma & Crosato 1989).

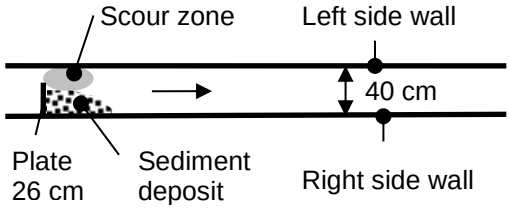

Figure 3. Flume with a transverse plate.

After reaching an equilibrium bed configuration (Figure 4) in the base case, two locations are selected for the starting point of the longitudinal training wall in the next experiments. The training wall is 1,7 m long and is placed at a distance of 10 cm from the glass side wall, obtaining two parallel channels, one 30 cm wide and the other (side channel) 10 cm wide.

Data collection and processing

Data collection

Bed level and water level are recorded by 5 laser devices, i.e. laser 1 measures the water level, laser 2 to laser 5 measure the bed level. Lasers are put transversally to the flow from the right to the left side wall (Figure 5). Data are measured three times a day: in the morning, at about 9 a.m., around mid-day and in the late afternoon, around 5 p.m.

Figure 4. Equilibrium state of bar mode = 1 on the experimental flume.

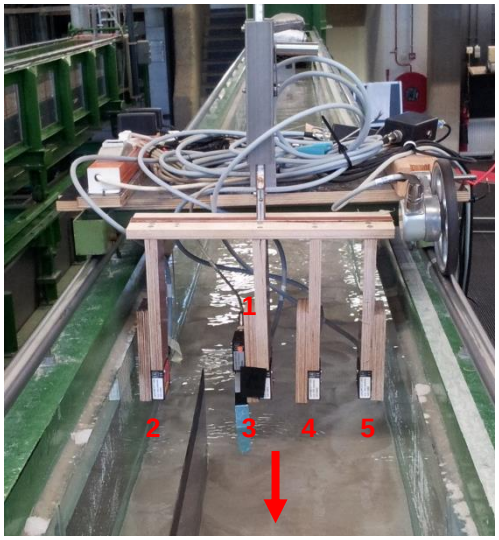

Figure 5. Laser devices to measure bed level. Laser 1 is in upper part, laser 2 to 5 are from the right side to the left side.

Velocity is measured by PTV method each time the bed level is measured. From these PTV measurements, a velocity field can be reconstructed and the cross-sectional variation of flow velocity is derived from it.

Data processing

Due to the presence of relatively large dunes and ripples (Figure 4), the rough data are filtered to clean out the bar signal. On this study, the filter used is based on the Matlab software ProcessV3 and optimized for bed forms having wavelength larger than 1 m. Figure 6 shows a typical raw signal (in green) and the filtered longitudinal bed topographies (in red). Unfortunately, the filtering procedure also reduces the bar amplitude.

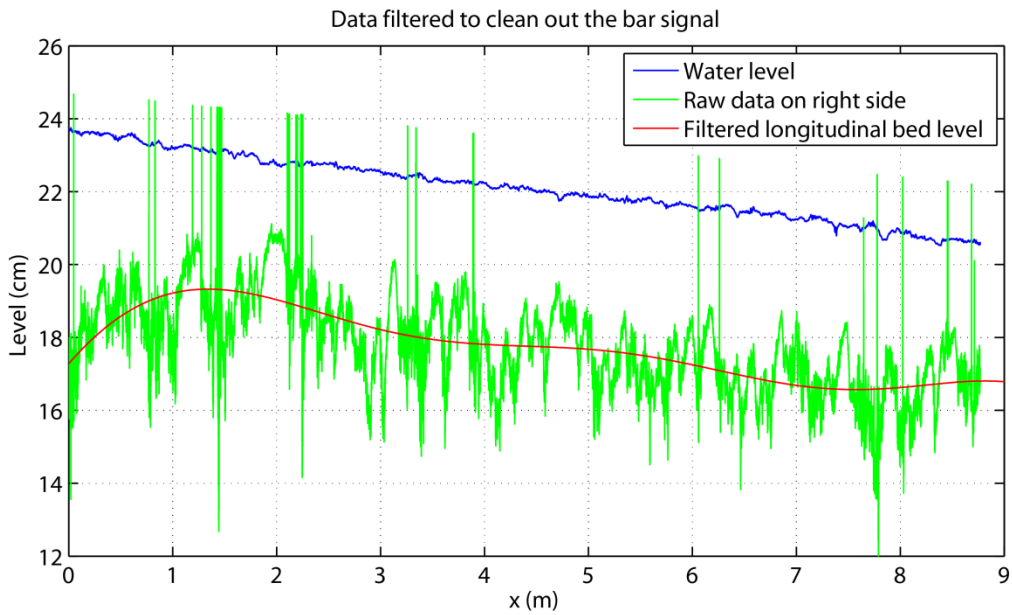

Figure 6. Data filtered to clean out the bar signal. Raw signal (green); longitudinal bed level profile 5 cm from the right side wall (red); water level (blue).

Preliminary results

Reference case

Steady alternate bars started to develop immediately after experiment start, forced by the presence of the plate obstructing the flow, which caused local scour opposite to a large sand deposit (Figure 3). These bars grew up and became more pronounced after some time. However, without sediment feeding the bed level degraded quickly and it became difficult to identify the bars. For this reason, the bed topography after 48 hours of development was used as reference situation. In this case the steady alternate bars with a wavelength of 2.5 m and amplitude of 1 cm were well recognized (Figure 7). Based on this, the following locations of the starting points of the training wall were selected: position D, near the bar top (Figure 2), and position A1, at the upstream start of the bar (Figure 2), 4.4 m and 5.5 m from upstream boundary, respectively. These two positions were expected to lead to two different developments of side and main channel as predicted by Le et al. (2014) (Figure 2).

Training wall starting at D

With the training wall starting at D, the bed level in the side channel became lower than in main channel (Figure 8). The difference in bed levels between the two channels increased with time (Table 1). At the end of day 3, the difference was 1.1 cm, mainly due to scour in the side channel.

With the same water level in the main and side channels, this result means that the water depth in the side channel became larger than the one in the main channel. In addition, the flow velocity in the side channel was larger than the flow velocity in the main channel (Figure 9).

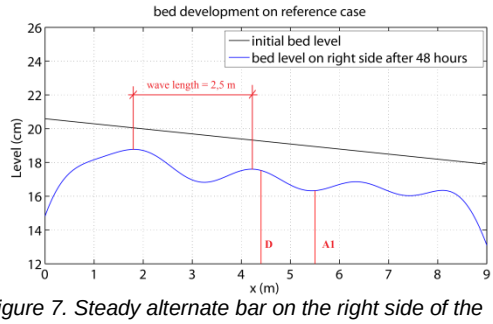

Figure 7. Steady alternate bar on the right side of the flume in reference case. Positon D was on downstream part of a bar and position A1 was on upstream part of another bar.

Although the duration time of this test was only 3 days, this result shows a clear trend that erosion in the side channel develops confirming the predictions made by Le et al (2014).

Table 1. Difference on bed level between main and side channel when training wall starts at D.

Time step	Main channel level (cm)	Side channel level (cm)	ΔZ (cm)
Day 1	18.0 ÷ 17.4	17.7 ÷ 17.3	0.3 ÷ 0.1
Day 2	17.3 ÷ 16.9	16.9 ÷ 16.6	0.4 ÷ 0.3
Day 3	16.7 ÷ 16.4	16.0 ÷ 15.3	0.7 ÷ 1.1

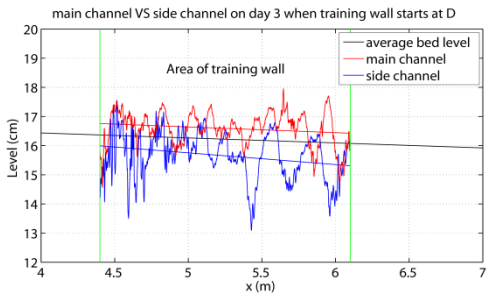

Figure 8. Bed level in side channel was lower than in main channel when training wall starts at D.

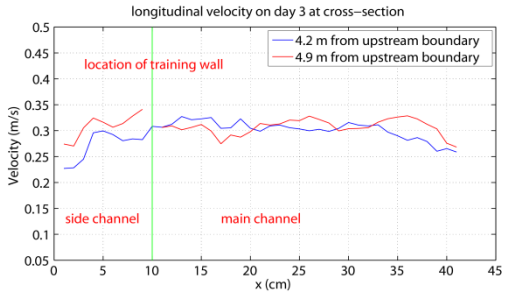

Figure 9. Longitudinal velocity in side channel was higher than in main channel when training wall starts at D.

Training wall starting at A1

For the case of a training wall starting at A1, Le et al. (2014) predicted aggradation in the side channel compared to the main channel. However, during the experiments, the discharge varied between 4.0 l/s to 5.0 l/s due to unexpected rising of pressure on pumping system. This altered the bars and therefore the results of this test are unreliable (Figure 10).

Table 2. Difference on bed level between main and side channel when training wall starts at A1.

Time step	Main channel level (cm)	Side channel level (cm)	ΔZ (cm)
Day 1	17.63 ÷ 17.10	17.56 ÷ 17.12	0.06 ÷ (-0.02)
Day 2	16.98 ÷ 16.67	16.83 ÷ 16.57	0.15 ÷ 0.10

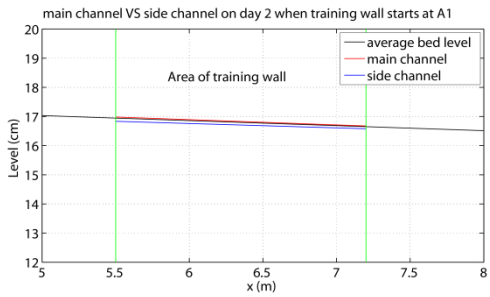

Figure 10. Bed level in side channel was a little bit lower than in main channel when training wall starts at A1.

Future work

The work is on-going and this preliminary work will be followed by extensive experimental investigations to carefully verify the long-term effects of a longitudinal training wall in a river with alternate bars, starting at different locations related to a bar. Several configurations will be studied, characterised by discharge, sediment, width of the side channel, length of the training wall and presence of opening points on the training wall.

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Observed evolution of side channels

R.P. van Denderen*[1], R.M.J. Schielen[1,2], S.J.M.H. Hulscher[1]
[1] *University of Twente, Water Engineering & Management, P.O. Box 217, 7500 AE, Enschede, the Netherlands*
[2] *Rijkswaterstaat, Water, Traffic and Environment, P.O. Box 17, 8200 AA Lelystad, the Netherlands*
** Corresponding author; e-mail: r.p.vandenderen@utwente.nl*

Introduction and physical processes

Side channels are a common natural feature of rivers, but side channels disappeared in European and North-American rivers due to river regulation. Anthropogenic adjustments led to several changes in the river planform, for example, the narrowing of the channel to increase the depth during base flow, the straightening of the river by bend cut-offs and the reduction of floodplains to increase the agricultural benefit. These adjustments led to an abandonment of side channels which increases flood levels, changes the morphological equilibrium and reduces the ecological value of the river. Nowadays, river managers use side channels to reduce the flood levels and to increase the ecological value of the river and regular maintenance is required due to unexpected erosion and aggradation in the channels. In order to reduce the maintenance costs the morphological processes, which influence the evolution of side channels, have to be better understood.

A side channel consists of three main elements: a bifurcation, a confluence and a system of two channels. Each of these elements are important for the evolution of side channels. The morphological evolution of a bifurcation and the distribution of the sediment over the branches, depends on the local flow patterns. In Fig. 1 a schematization of the flow patterns at a bifurcation is presented. Depending on the angle of the bifurcation and the flow velocity, a flow circulation zone is formed in the side channel. In this zone the flow velocities are lower which results in aggradation. On the other side of the side channel an increased flow velocity causes scour and bank erosion. Other more detailed processes are hidden below the water surface and not visible from aerial photographs which are studied here. At confluence similar flow patterns occur as shown in Fig. 2. Depending on the angle of the confluence a flow circulation zone is formed in the downstream channel with the corresponding aggradation and on the other side of the channel the flow accelerates which results in erosion. At the location where the two flows meet, a shallow zone is formed due to the lower velocities. Best (1987) called this zone the stagnation zone.

The characteristics of the main channel and the side channel influence the discharge distribution and therefore also the morphology. If a large amount of discharge is diverted into the side channel, erosion in the side channel and sedimentation in the main channel is likely to occur. However, if the flow velocity in the side channel is too low for sediment transport, this will lead to the closure of the side channel. The characteristics of the main and side channel depend on the objectives of the river managers. An ecological side channel conveys generally less discharge than a side channel which is designed to lower the flood levels. Moreover, when the river has an important navigational function, the discharge in the side channel will be minimal to prevent a bed level increase in the main channel.

In this abstract a few examples of side channels are presented. Each of them show characteristic behaviour in relation to the theory presented above. The goal is to show the influence of several side channel characteristics on its morphological evolution.

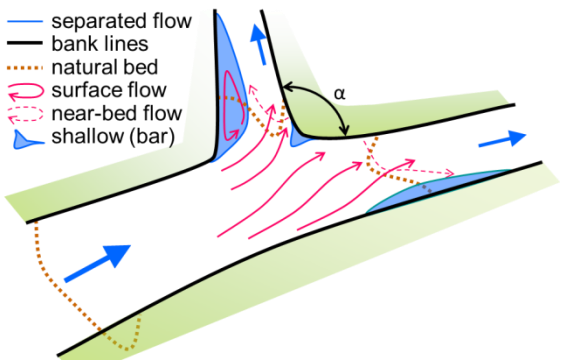

Figure 1. Flow patterns at a bifurcation after Kleinhans (2013) .

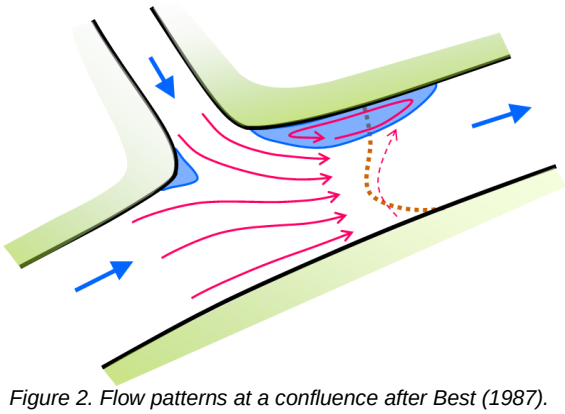

Figure 2. Flow patterns at a confluence after Best (1987).

Bifurcation angle

As shown in Fig. 1, the shape of the bifurcation has an influence on the flow pattern at bifurcation. One of the main parameters is the bifurcation angle. This angle influences the discharge and sediment partitioning, and also influences the size of the flow circulation zone. Large bifurcation angles occur in nature when the river banks are strong compared to the flow (Kleinhans, 2013). Otherwise the increased flow velocity, due to the reduced inflow width of the channel, causes bank erosion. Fig. 3 presents the evolution of a side channel in the river Drau (Austria) just after construction in 2002. In 2002 the bifurcation angle was almost 90°, but in 2003 the angle reduced to about 45°. At the same time a bar was formed in the flow separation zone as was expected from Fig.1. The flow was strong enough to erode the banks which allowed for the reduction of the bifurcation angle and prevented the bar from closing the channel entrance.

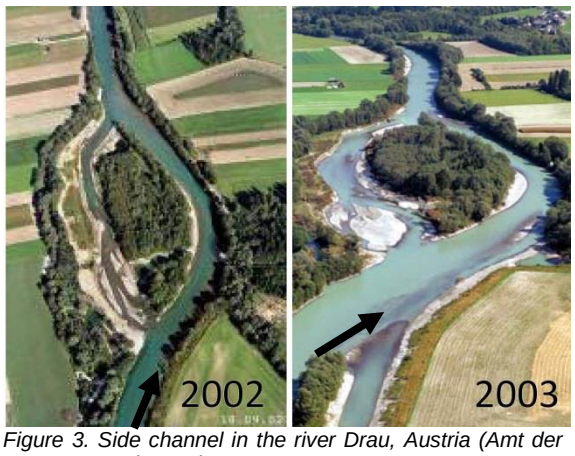

Figure 3. Side channel in the river Drau, Austria (Amt der Kärntner Landesreg.).

Constantine et al. (2010) present other examples where the bifurcation angles are large but the discharge in the side channel is relatively small. Similar to the Drau, a bar is formed at the entrance of the channel, but due to the low flow velocity this bar closes the side channel completely. Dieras et al. (2013) studied a side channel of the river Ain (France) with a small bifurcation angle. The channel was closed due to the low flow velocity, but the aggradation was more evenly distributed over the channel. The bifurcation angle has therefore an influence on the mechanism which causes the side channel to close.

Confluence's stagnation zone

Fig. 2 shows that at a confluence several flow patterns can lead to morphological changes. These changes do not always have an influence on the evolution of the side channel, but can have negative effects on for example the navigability of the river. Fig. 4 shows an example of a side channel which closed at the outflow. In photo of 1996, a large bar is present near the outflow of the side channel which corresponds to the stagnation zone in Fig. 2. The discharge in the side channel is small which allows the bar to close the side channel completely. The flow velocities in the side channel are insufficient to erode the bar. After a few years, satellite images show that the bar was breached which is most likely caused by a flood event.

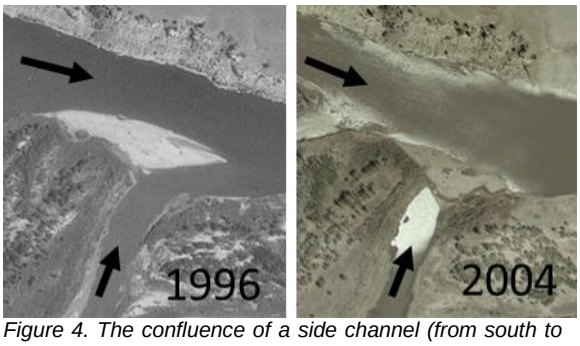

Figure 4. The confluence of a side channel (from south to north) with the main channel (from west to east) in the Yellowstone River, Minnesota USA (Google Earth).

Side channel length

The morphological evolution of the side channel and the main channel are strongly dependent on the hydrodynamic conditions in the channels. A large discharge in the side channel can lead to erosion in the side channel and a reduced discharge in the main channel often leads to aggradation. An important parameter is the length of the side channel relative to the main channel length. A relatively large side channel length results in a relatively larger discharge due to the larger slope which often causes erosion in the side channel and sedimentation in the main channel.

An example of a side channel which is very long compared to the main channel is shown in Fig. 5. This side channel is connected to the river Elbe, upstream of Magdeburg, Germany. This old river bend was originally closed to reduce the length of the main channel for navigation benefits. In 2001 the channel was reconnected upstream to increase the connectivity and to promote the ecological restoration of the river. However, the side channel is 2.5 times longer than the main channel and the width of the side channel suddenly increases. This results in a small discharge in the side channel and a deceleration just after the entrance of the channel which causes a large sedimentation. Therefore in a relatively long side channels the flow velocities are too small to transport sediment and to prevent the close of the side channel.

Figure 5. The morphological evolution of the side channel at Kurzer Wurf in the river Elbe (Germany) after its opening in 2001 (Google Earth).

Side channels which are shorter than the main channel are found in the inner bends of rivers. Due to the shorter channel, the slope is relatively large and the channel will therefore attract relatively more discharge, which can lead to large erosion in the side channel. In an extreme case, if the erosion in the side channel is too large, it can cut-off the old main channel which in many rivers is an unwanted effect. Therefore in relatively short side channels obstructions/structures are placed which reduce the discharge and the erosion in the side channel. Fig. 6 shows a side channel in the Trinity River in the USA. This side channel was built for ecological purposes and the length of the side channel is about 10% shorter than the main channel. To reduce the discharge in the side channel, tree logs were lain into the stream which are keyed into the bank and ballasted by boulders. These obstacles increase the roughness and reduce the discharge in the side channel. However, from the photo of 2011 it is possible to conclude that too many obstacles were placed since large sedimentation occurs in the downstream part of the side channel. This could suggest a closure of the side channel in the future.

Figure 6. Side channel in the Trinity River, California USA (Trinity River restoration program).

The amount of discharge which is diverted to the side channel has a large influence on the morphological evolution of the side channel. The slope of the side channel relative to the main channel is an important parameter. By using obstacles/structures the discharge can be regulated, but it not always obvious how the morphological evolution is affected by these structures.

The objectives of river managers

The construction of side channels can have different objectives. The most common objectives are to recreate habitat and to reduce the flood levels during high water. In this section several side channel are discussed which were constructed with different goals.

The side channel in the river Drau (Austria) which is shown in Fig. 3, was constructed to increase the ecological value, to reduce the flood risk and to recreate morphological processes (Formann et al., 2007). Before its canalization in the 1960s, the Drau was a braided river with large gravel bars. Measurements show that since the construction of the side channel, gravel bars have formed in the side channel and main channel. This suggests that due to the construction of the side channel, the morphological variability has locally been restored.

Another reason for constructing a side channel is to reduce erosion in the main channel. To benefit the navigability, rivers were canalized resulting in the reduction of the cross-sectional area and the erosion of the main channel. This erosion continues, but the amount of erosion differs over the river due to, for example, armoured layers which results in bed level steps in the main channel. The bed level steps are now an obstacle for the navigation during base flow conditions. To mitigate these effects, side channels can be constructed which increase the cross-sectional profile and thereby reduce the erosion in the

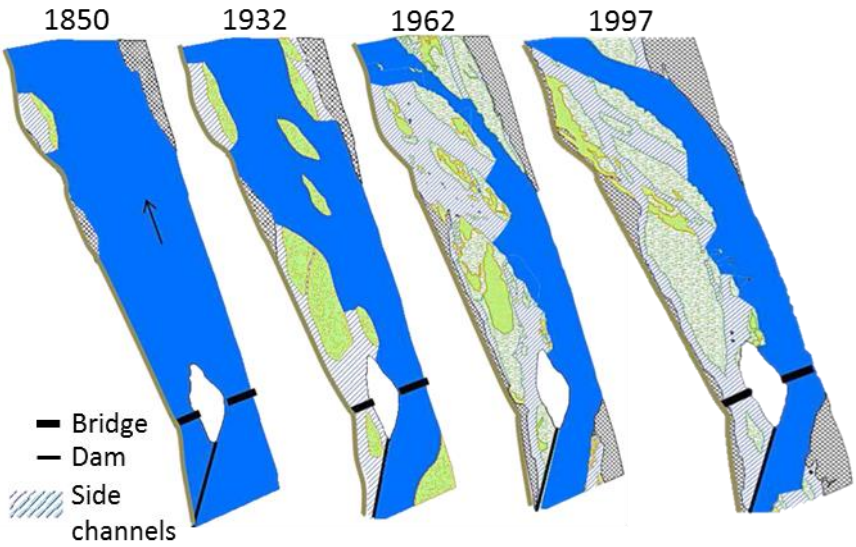

Figure 7. The formation of side channels at Charité-sur-Loire in the river Loire (France). (After Nabet (2014))

main channel. Side channels which were built for this reason are for example at Regelsbrunn in the Danube (Austria) and at Rees in the Rhine (Germany). In both cases the cross-sectional area was originally reduced to benefit the navigability, but side channels were recently (re)created to increase the cross-sectional area.

The reduction of the cross-sectional area does not always lead to the disappearance of side channels. At Charité-sur-Loire (France) a dam was built in 1850 to reduce the cross-sectional area and to induce erosion in the main channel. This dam is shown in Fig. 7. Since 1850 large erosion on the right side of the river has increased the navigability, but due to the dam, the left side of the river aggraded. The dam overflows during high water levels and therefore secondary channels were formed behind the dam. This shows that side channels can also be consequence of human interference.

Discussion and conclusion

The abandonment of side channels is caused by a combination of hydrodynamic and morphologic conditions. Here, several cases were presented to show the influence of several parameters on the morphological evolution in the side channels. However, the usage of satellite/aerial images is limited since the bed level changes and morphological processes below the water surface are hidden. The next step is therefore to look in more detail to certain processes using detailed measurements.

From this analysis it becomes clear that characteristics of the bifurcation, confluence and the two channels have a large influence on the morphological evolution of the system. Another important parameter are the goals of the river managers. These goals define the main characteristics of the side channels.

Acknowledgments

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Bridging gaps by internalizing end users challenges and market needs

A.G.A. Schellekens and D.W. van Raalten
ARCADIS The Netherlands, Amersfoort office, Eric.Schellekens@arcadis.nl, David.vanRaalten@arcadis.nl,

Introduction

Both applied science and fundamental science could or should have value added. But as we all might know and have experienced alignment between on the one side end-users willing to pay, consultants integrating state-of-the-art knowledge into their advice, construction companies willing to apply and on the other hand the scientific community lacks in many cases. For us, consultants, bridging the gaps between wonderful research, insights and the needs regarding world trends and market demands is core (or at least needs to be core).

In this keynote session we'll share proven ways and insights to align research and market, to align research and 'bankability', to raise competences, to widen scopes of thinking and acting. We take you with us on our journey presenting key factors of 'business modelling' (e.g. mains of the CANVAS model, Osterwalder et al., 2010), presenting a promising pathway to assess and raise institutional capacity (e.g. CAP4PE, formerly known as PACT, Ballard et al., 2013), presenting a pathway to accelerate city and area transformations (the RP2.0, Schellekens, et al, 2015), presenting needs from world leading construction and dredging companies.

Of course to discuss and to end up with 7 one-liners to support bridging the gaps together.

To engage all of you and to provide foods for thought we put below one of our 'hangers' to construct our consultancy around, The Resilience Pathway2.0 (RP2.0). A pathway focused on implementation of solutions, while using high level solutions, adaptive stakeholder management methods and using frontrunner knowledge, expertise and experiences beyond the core 'technical' scope.

Method

There is an urgent need for climate adaptation and for other either social of biophysical stressors to avoid huge socio-economic losses (growing from today's $6bn to $60bn p.a. by 2050 in the 136 largest coastal cities alone, Hallegatte et al., 2013). However, the lack of good pathways to focus on real implementation beyond policy integration, non-availability of money and gaps in capabilities are huge barriers. A flow of private money, either besides or apart from public money, needs to be unlocked if adaptation is to happen quickly enough (Hinkel et al., 2014, IPCC, 2014). Research (ABL / Ricardo AEA, 2013, Defra, 2013) shows that many cities (around 80% in Europe) have low or very low capacity and therefore cannot specify required resilience measures.

To unlock money flows and to raise capacity of stakeholders, we recently developed and successfully tested a new and unique high level business model, the '**Resilience Pathway 2.0**' (figure 1), with European cities and reached out to potential partners, with encouraging results.

Figure 1. The Resilience Pathway and its building blocks

Existing high level engineering skills and products and services have been brought together in partnership with leading deal-structuring experts and financial organisations to help riverine and coastal cities to access private and public money for transformation (e.g. climate adaptation).

Resilience is most needed for long-lasting decisions with outcomes that last 10+ years (e.g. water and other infrastructure), which need to be viable in unpredictable and fast changing future climate, energy and demographic scenarios, where the status quo is usually the least likely scenario. These decisions are difficult and / or expensive to reverse (in money and energy terms). This is where identifying 'moments of change' and defining 'investment opportunities' seem the way to go, while creating a 'marriage' between technical, financial and social engineering.

Results

The main phases of the Resilience Pathway 2.0 have been explained in the text below.

Phase 1: Scoping

- The output of this first phase is a clear briefing document clarifying the challenges to be addressed, also to identify potential opportunities ('moments of change') to address them.
- Area dynamics and ambitions. The 'area dynamics' aspect involves identifying known and likely activity around which 'moments of change' might be constructed. It is also important to identify the city's wider agenda and ambitions.
- Climate challenges and other stressors. This building block assesses the vulnerability of an area or city arising from changes in extreme weather events and from climate changes in both the short and the long term. It also defines mitigation and other challenges that result of social and environmental stressors.
- Capacity and Framework assessment. The 'capacity' element of this building block aims to define capability gaps that need to be filled to implement Resilience Pathway 2.0, also to identify framework conditions (legislation or regulation, incentives, etc., to be taken into account in the 'optioneering' phase (Figure 2).

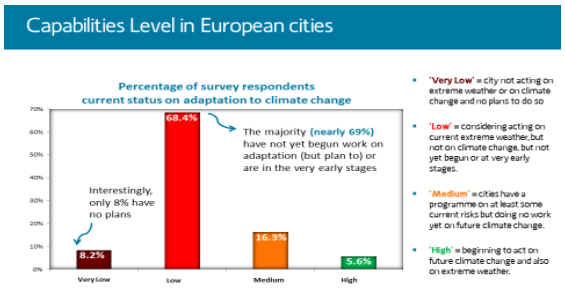

Figure 2 ASEC survey on capacity levels in 196 European cities (Ricardo AEA and Alexander Ballard Ltd., 2013)

Phase 2: Optioneering

The purpose of the Optioneering phase is to identify one or several promising, fundable and feasible business cases around a development (or a combination of developments). These will provide alternative routes to meeting climate and other objectives of projects that will be attractive to private and other investors. This is a unique and crucial phase, where bringing high-level engineering and financial thinking together is essential in unlocking finance for resilience.

- High level solutions. This involves defining and designing at a high engineering level the development in a way that is profitable and fundable, while also meeting the ambitions and respecting the parameters identified during the scoping phase.
- Socio economic Cost Benefit Analysis. Fundamental to fundable and profitable solutions s identifying socio-economic losses and benefits from the various options.
- Financial and legal arrangements. The third building block is to develop suitable financial arrangements for promising options, to test whether these financial arrangements align with the interest of investors and also meet the needs of clients and to reduce risks and to apply an adaptive stakeholder management.

Phase 3: Deal Structuring

The purpose of the third phase, the Deal structuring phase, is to transform the selected intervention opportunity – as identified in the Optioneering phase – into a contract with detailed specifications and financial and legal arrangements (e.g. permitting') to deliver the project objectives: resilient projects with an acceptable (typically low) risk-return profile. Again, there are three building blocks.

- Detailed solution engineering and permitting. The first building block focuses on transforming the high-level engineering approach identified during the 'optioneering' phase into solutions engineered in detail, aligned with the permit conditions that have been negotiated.
- Detailed financial engineering. The second building block focuses on transforming the high level financial arrangements chosen during the 'Optioneering' phase into detailed financial arrangements. For example, 'Special Purpose Vehicle' companies need to be set up, detailed contracts signed, detailed due diligence completed.
- Specifications and strategic control. The third building block elaborates the specifications of the promising opportunities in a detailed way, e.g. to allow sub-contracts to be let, i.e. aligned with the contract requirements and wishes. The strategic control aspect focuses on how to avoid collapsing goals.

Phase 4: Project implementation

The aim of the project implementation phase is to get the projects or developments realised and well managed. This phase is a missing link in the initial focus of many strategies.

- Project realisation. The first building block is about getting the development realised. This will sometimes differ from traditional project realisation in that climate adaptation or climate mitigation or both are seen as one of the main goals of the project or development. Also, more projects will have a 'breakthrough' element.
- Asset management. The second building block is about the maintenance of the assets of the project or development in order to optimize the climate impact (and other desired outputs) in the long term as well as over the life of the project.
- Learning and Capacity and Framework development. The third building block focuses on creating the projects as 'learning laboratories', designed to raise the effectiveness of future interventions. In addition, capacity gaps identified in the Scoping phase will be addressed l (e.g. by education, coaching, hiring, etc.).

Conclusion

To adapt in time to extreme weather events, the gradual climate change and other stressors in order to avoid huge costs and inefficiency either now or in the future, planning strategies need to focus on implementation. Strategies that align the needs of all stakeholders, both representing the demand side and the supply side, e.g. the needs of authorities, companies, project developers, big industries, investors and insurers. To unlock money flows and to raise capacity of stakeholders, ARCADIS has developed and successfully tested a new and unique high level business model, the 'Resilience Pathway 2.0', and .reached out to potential financial partners, with encouraging results. Existing high level engineering skills and products and services have been brought together in partnership with leading deal-structuring experts and financial organisations to help riverine and coastal cities to access private and public money for climate adaptation.

The Resilience Pathway. A pathway to be seen as a new of master planning and city and area development, with both promising impacts, business and research opportunities. So "Think Moments of change, Act high level and Talk revenues while 'optioneering'.

The new and holistic way of thinking behind the Resilience Pathway 2.0, capacity building and business modelling, makes innovations and new knowledge scalable around the world. Combined with insights in the concrete needs of huge and world leading companies and end users research, both applied and fundamental, could be focused on being applied in the short or long term.

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3 – Poster abstracts

Development of a new research tool for modelling the long term 2D effects of sediment management measures

L. Arkesteijn[1*], A. Blom[1], R.J. Labeur[1]
[1]Delft University of Technology, Faculty of Civil Engineering and Geosciences, Department of Hydraulic Engineering, P.O. Box 5048, 2600 GA, Delft, the Netherlands
** Corresponding author; e-mail: e.c.m.m.arkesteijn@tudelft.nl*

Introduction

Many normalization works have been conducted in large rivers with the objective to increase safety, ease navigation, generate hydropower or control water supply. Such large human interventions in the river system naturally come with a number of side-effects. Some of these can be (temporarily) resolved by implementing sediment management measures. For instance, the construction of a dam typically causes sediment to be trapped in the upstream reservoir, resulting in a reservoir that needs to be dredged, flushed, or bypassed, and a deficit of sediment downstream of the dam that can be compensated by applying nourishments.

Historically, the design of such measures was based on trial-and-error, by using some rules-of-thumb or simply by looking at the available funds. However, this has changed in the past few decades, and now numerical models are more frequently used to predict the river's response to a particular measure and thereby aid in the optimization process (e.g. Paarlberg et al., 2015). A number of tools is readily available, but these tools have shortcomings in the sense that they are either 1) only one dimensional (e.g. Viparelli et al. 2011), 2) not suited for long term computations (100-1000 years) in combination with large domains (e.g. DVR, Sloff, 2011, and DredgeSim, Maerker and Malcherek, 2010), or 3) have no implementation of mixed size sediment processes (e.g. Miwa and Parker (2012)). The latter is important since mixed size sediment processes can have a significant influence on the effects of sediment management measures.

During a research project that is carried out within the scope of the Dutch research programme RiverCare, we will develop a new research tool that does not have the shortcomings listed above. The main concession that we will make to enable this, is a simplified flow modelling which is aimed to be only valid in combination with morphodynamic changes on the long term. In this abstract we will first specify the need for this new tool, after that discuss the research methodology and finally introduce some of the cases we intend to use for validation and application.

A new research tool

The shortcomings that we are trying to overcome with this new research tool are focussed around the inclusion of two dimensional mixed-size sediment processes. One dimensional models are suitable in many situations, especially where lateral variations of river parameters are limited. However, for predicting the effects of sediment management measures this is often not sufficient. Dredging, for instance, is frequently performed in inner bends and dumping of sediments is often scheduled in outer bends, in order not to hinder navigation. Secondary flow and differences in flow velocities between inner and outer bends causes sediment to especially settle in the inner bend. Therefore, to predict future dredging needs, at least a two dimensional model is necessary.

The second shortcoming of existing tools is the ability to simulate long term effects. Most of the existing research tools that allow for a two dimensional estimation of river morphodynamics are computationally too demanding to simulate periods longer than approximately 50 years. Yet morphodynamic changes are typically slow and the interest here is especially in the long term effects of the sediment management measures. Hence the tool that will be developed needs to be able to make long term predictions.

The last issue is the inclusion of mixed-size sediment processes. By considering only unisized sediment, processes such as grain-size selective transport, vertical sorting and bend sorting are ignored. For sediment nourishments, however, these processes are crucial. Gölz et al. (2006) analysed a nourishment tracer experiment conducted in the Rhine, downstream of the Iffezheim dam, and clearly observed that the downstream propagation speed of the nourished material depends on its grain size. In addition, Berkhout (2015) showed that also vertical sorting is

important in the context of the effects of nourishments. He showed that coarse nourishments can cause bed degradation downstream. A numerical tool for such cases therefore needs to include these mixed-sediment mechanisms.

Finally, it is good to note why we do not try to adjust or complement an existing tool. Suitable candidates for this would be two dimensional mixed-size sediment models, since then only the computational time needs to be decreased. The problem with this approach is the structure of such models. They are usually numerical modules that are part of larger, rather complex model packages, and coupled to other modules for hydrodynamics and morphodynamics (see figure 1).

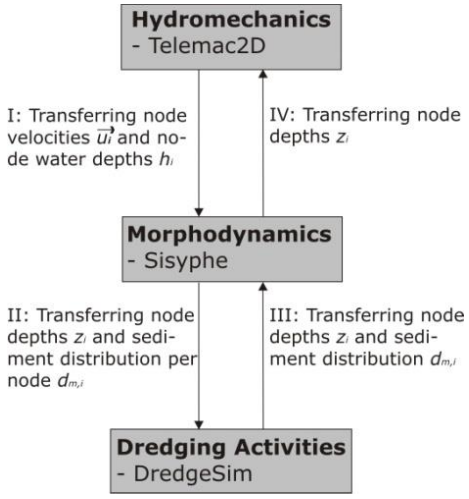

Figure 5: Modular structure of DredgeSim tool (Source: Maerker and Malcherek (2010)).

Typically the core of such model packages is the hydrodynamics module, which, combined with other modules, is applicable in very diverse problems. The main drawback of this general applicability is that processes that might be negligible in a specific application, can unnecessarily complicate and prolong the computations for that application. This is likely to be the case in our situation, where fast and small scale fluctuations in flow variables are probably of negligible influence to morphodynamic changes on the large morphodynamic time scale. More specifically, it may in this case be justified to consider quasi-steady flow without advective accelerations, greatly simplifying the equations and thereby reducing the computational time. Since this involves drastic changes of the equations that the module is based on, it is easier to construct a new hydrodynamic module instead.

This still leaves the option of using an existing morphodynamic module. However, these modules are usually based on the Hirano model for which there is a known issue with ellipticity of the system of equations (Ribberink (1987)). We will start by implementing the Hirano model, but also consider other descriptions of mixed size sediment mass conservation.

Research methodology

The research can roughly be divided in two parts; the development of the numerical research tool, and the validation and application of the tool to a number of test cases.

For the development of the tool we will start with an analysis of the system of equations. As stated before, most models are composed of separate modules for hydrodynamics and morphodynamics. We will first investigate the hypothesis whether these modules can be decoupled (Cao and Carling, 2006). If that is the case, we will continue by analysing the flow and sediment equations separately. We will verify whether the quasi-steady approach and neglecting the advective term are justified for our application, and also look for other opportunities to simplify the equations that allow for a more efficient solution strategy.

Alongside the search for simplified equations we will focus on a dedicated numerical algorithm to solve the system of equations. We will implement the simplified flow equations and the Hirano equations for sediment mass conservation and try different solution strategies. Initially, this will be done for only one dimension, however, the strategies should allow for extension to two dimensions later on. After that, these modules will be adjusted (e.g. the extension to two dimensions), made more efficient (e.g. a different numerical technique) or replaced (e.g. a sediment mass conservation equation not based on Hirano) in an iterative process. After each adaptation the numerical properties are checked and the model is validated using some problems with known analytical solutions.

Finally, a sediment management module that contains the possibility to simulate dredging and nourishment activities will be coupled to the tool. The tool can then be validated and applied to flume experiments and field cases during the second part of the research.

Validation and application

The following test cases will be used for validation and application of the research tool.

Flume experiments
Flume experiments are crucial for the validation of a model, since all measurements are taken in a more or less controlled environment. During this research project we aim to use the flume data gathered by Berkhout (2015) for validation of the model. The dataset describes the propagation of a shoal using both unisize and mixed-sized sediment. This makes it especially suited for the validation of the morphodynamic module and to test different approaches for the sediment mass conservation equation.

The above experiment was conducted in a straight flume and therefore mainly contains one dimensional processes. We intend to perform another flume experiment within this research that focuses on two dimensional mixed-size sediment processes. This would greatly benefit the validation of the modelling of the two dimensional processes, since also in the field cases the availability of data in more than one dimension is sparse.

Iffezheim tracer experiment
In Iffezheim a nourishment of 170.000 m^3/a was started at April 18th 1978, and is continued up until now. The motivation for the nourishment was to supplement the deficit of sediments caused by the construction of a dam in Iffezheim at 1978. Data on water level elevation, cross-sections and discharges are available for both the pre-dam and post-dam situations. Based on these, the nourishment was claimed successful in 1982, however the exact effects were still rather unknown. In 1990 a tracer experiment was initiated to investigate the effects in more detail. Samples of the bed grain-size in a surface and subsurface layer were sampled during a period of five years. Gölz et al. (2006) analysed the results, and in figure 2 the propagation speed of the nourished material is shown.

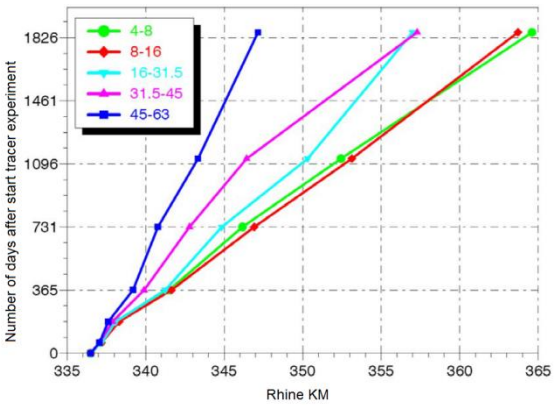

Figure 2: Propagation of the center of mass per grain size fraction (Source: Golz et al. (2006).

The different propagation speeds of each grain size emphasize the need for a model that can incorporate mixed-size sediment processes. The propagation speed of each grain size fraction, bed elevation profiles and also the (limited) vertical mixing data will be used for the validation of the research tool. As far as we know, this is the only field dataset on the effects of sediment management measures in combination with mixed-size sediment and in addition, it is also a data set describing a relatively long period. Therefore it will be very important for the validation of the newly developed numerical tool.

Dredging case study
Rivers are dredged to ensure navigability, usually on a performance based contract. In many rivers there is a requirement that dredged material should be returned to the river. Different strategies for this include variations in dredging over-depth (clearance) and variations in dumping locations. Recently, Paarlberg et al. (2015) used the DVR module of the Delft3D toolbox to test some 'smart dumping' strategies in the Parana River that would ideally decrease the amount of material that needs to be dredged on a yearly basis. In this research we will try to have a similar case study, either using dredging data from the Parana River or the Rhine-branch Waal. For both rivers, relatively large datasets are available, allowing to validate the model's long term behaviour.

Dutch nourishment pilot Lobith
The Dutch part of the Rhine suffers from ongoing bed degradation. Rijkswaterstaat will start a large scale field experiment in the Upper-Rhine in 2016 to study whether nourishments can be used to counteract this. During the experiment measurements will be taken that can be used to validate the model. With this nourishment pilot we aim to not only validate our model, but also to use it to make predictions for the long term river response investigating the effects of the nourishment in general, and more specifically on the Pannerdense Kop. In addition, the validated morphodynamic model could also help us to better understand the causes of the bed degradation in the Dutch part of the Rhine.

Conclusion
We have presented a research plan for the development of a numerical research tool that can be used to model the effects of sediment management measures. The need for a new tool, the research methodology, and cases for application have been discussed.

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Transverse bed slope experiments in an annular flume

A.W. Baar, J.C. De Smit, M.G. Kleinhans
Faculty of Geosciences, University of Utrecht, PO Box 80115, 3508 TC, Utrecht, the Netherlands

Introduction

A crucial part of morphodynamic models is the transverse bed slope effect, which determines the deflection of sediment transport on a transverse sloping bed due to gravity. Overestimating this effect leads to flattening of the morphology, while underestimating leads to unrealistic steep bars and river banks (Fig. 1). Therefore, incorrectly estimating the transverse bed slope effect could also have major consequences for the predicted large-scale morphology, as it influences the development of river bifurcations, meander wave length and the degree of braiding in rivers and estuaries.

In current models, the prediction of the magnitude of sediment transport is based on a situation of a flat bed with a single grain size, and only the direction is afterwards corrected for transverse gradients (e.g. Ikeda, 1982; Talmon et al., 1995). However, in reality sediment transport is also affected by grain size distribution, bedforms and suspension rate. The angle of sediment deflection is therefore often calibrated on measured morphology afterwards.

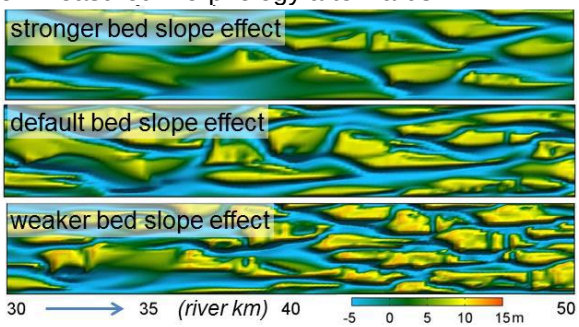

Figure 1. The effect of stronger and weaker transverse bed slope effect on channel morphology (Schuurman et al., 2013)

Previous research

Current transverse bed slope predictors are based on theoretical model studies (e.g. Odgaard, 1981; Sekine & Parker, 1992) and laboratory experiments. Experimental studies focused on experiments with bended flumes (e.g. Zimmerman & Kennedy, 1978; Struiksma et al., 1985; Ikeda & Nishimura, 1986), or straight flumes initiated with a transversely sloped bed that relaxed to a horizontal bed, to avoid secondary currents (e.g. Ikeda, 1982; Talmon et al., 1995; Talmon & Wieseman, 2006). However, in bended flumes the transverse bed slope effect cannot be isolated from the effect of helical flow, and in the experiments with a straight flume it is not possible to measure the equilibrium slope. Also, all previous transverse bed slope experiments were performed with a small range of flow conditions and mostly uniform grain sizes, varying from 0.09mm (e.g. Talmon & Wieseman, 2006) to 0.79mm (e.g. Talmon et al., 1995). Therefore, existing predictors can only apply for a certain range of conditions, as they depend on either bed load or suspension dominated conditions, and do not account for the presence of bedforms (Talmon et al., 1995, Wieseman et al., 2006). Odgaard (1981) also discusses that grain size and flow conditions vary along the transverse slope, and therefore understanding of sediment sorting patterns along a transverse slope is needed to accurately predict the transverse slope effect.

Aim and methodology

The aim of the current research is to experimentally quantify the bed slope effect for a large range of flow velocities, helical flow intensities and particle sizes (0.1-4mm), while taking into account the effect of bed forms and suspension. The experiments are being executed in the annular flume of Delft University of Technology (Fig. 2). This flume functions as an infinitely long bended flume, which therefore avoids boundary effects. Flow is generated by rotating the lid of the flume, which can be controlled to create a large range of flow conditions. Also, the intensity of the helical flow can be controlled by counter-rotating the bottom of the flume (Booij & Uijttewaal 1999).

The equilibrium transverse slope that develops during the experiments is a balance between the transverse bed slope effect, the bed shear stress caused by the helical flow and the centrifugal force caused by the rotation of the bottom of the flume (Fig.3). This balance depends on particle size, size range and density, and the rotation velocities of the lid and the bottom of the flume. All these parameters are systematically varied during the experiments in order to determine the separate effect on the equilibrium slope. Also, the effect of bedforms and suspension can be studied by the large range in flow conditions and grain sizes. For the first set of experiments only uniform sediment is being used. At a later stage, also poorly

sorted sediment will be used in order to focus on sediment sorting processes. By systematically varying all these parameters, we aim to develop a more physical based transverse bed slope predictor that is applicable in a wide range of model simulations.

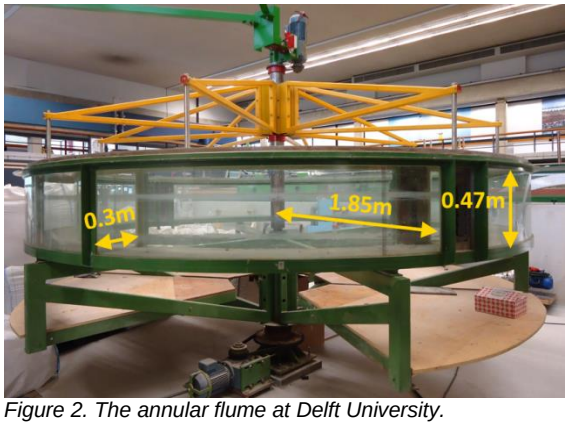

Figure 2. The annular flume at Delft University.

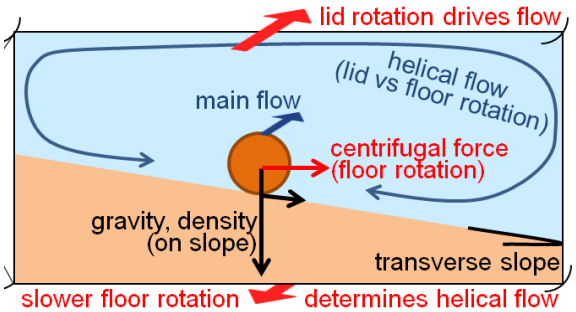

Figure 3. Forces acting on particles that can be controlled in the annular flume (after: Booij and Uijttewaal, 1999).

Preliminary results

In Fig 4. The morphology of several experiments with a grain size of 4 mm is visible, as well as the corresponding maximum transverse bed slope that developed. During this set of experiments the rotation velocity of the lid of the flume (ω_l) is kept constant, while the rotation of the bottom of the flume in the opposite direction (ω_b) increases. The average flow velocity therefore increases, as the difference between both rotation velocities increases. By increasing the bottom rotation, the centrifugal force on the sediment also increases, which counteracts the helical flow driving sediment inwards. Therefore, the transverse slope decreases with increasing centrifugal force, as is visible in Fig. 4B. When the centrifugal force is larger than the helical flow intensity, a steep slope develops towards the outside wall of the flume. Next to variations in transverse slopes, variations in dune length are also observed.

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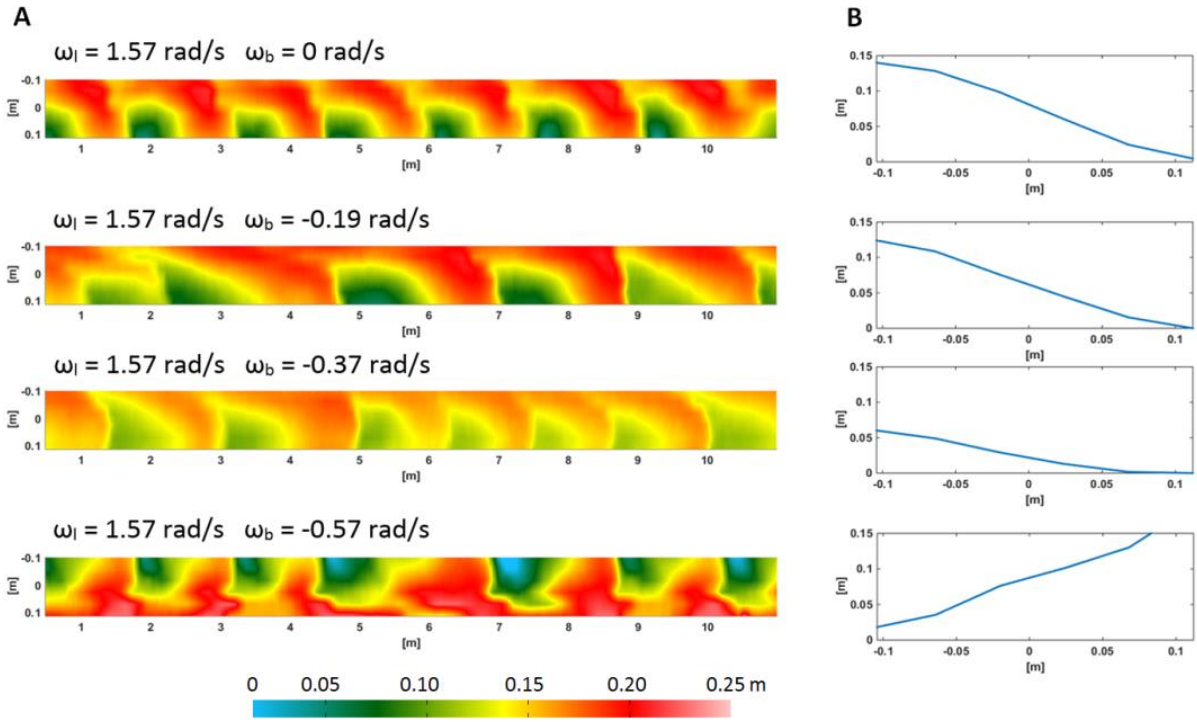

Figure 4. (A) Resulting morphology (top view) of a series of experiments with a constant rotation velocity of the lid of the flume (ω_l) and varying counter-rotating bottom velocities (ω_b), and (B) corresponding maximum transverse slopes. The width of the flume is measured relative to the average radius.

Do we have too much confidence in our models?

K.D. Berends* [1,2], J.J. Warmink[1], S.J.M.H. Hulscher[1]
[1] *University of Twente, Department of Water Engineering and Management, Faculty of Engineering Technology, P.O. Box 217, 7500 AE, Enschede, the Netherlands*
[2] *Deltares, Department of River Dynamics and Inland Water Transport, Postbus 177 2600 MH Delft*
** Corresponding author; e-mail: k.d.berends@utwente.nl*

Introduction

In 1999, the RIVM model credibility crisis (van der Sluijs 2002) forced modellers to think about validation and uncertainty of models. Models – especially numerical, physically based models of rivers – are ubiquitous in river engineering practice and river research alike. Although much progress has been made on model validation and uncertainty, it is our belief that an important facet of model use has received not enough attention.

Numerical modelling comes with its own particular set of problems. Because they are intangible compared to physical models, their credibility is largely derived from the authority of the governing equations and validity tests. However, there is no testing framework for one of the important uses for models, viz. situations that require the model to be modified from its calibrated state such as human interventions in the river system.

In this paper, we introduce our study into modified model validity and efficient uncertainty analysis.

Relevance of model validation

Driven by the expanding powers of computers, numerical modelling became very dominant in the practice of river hydraulic and hydrodynamics; both in science and professional consulting. Coincidental with the emanation of numerical models in the second half of the twentieth century, the paradigms in the philosophy of science changed as well. The classical, positivist view on science was renounced by Popper (Popper 1959) and replaced by two opposite philosophies (van Fraassen 1980): scientific realism (science produces true descriptions of the world) and constructive empiricism (science merely produces empirically adequate descriptions).

This fundamental dispute carried over to numerical modelling as well. It is commonly accepted that models applied in complex situations – not uncommon in river research – need some form of calibration. In hydraulics, some argue that the calibration exercise can be used to solve the inverse problem of finding the 'true parameter values' and that calibration becomes unnecessary once the true parameters are known or can be reasonably estimated (Abbott et al. 2001, Cunge 2003). Their imperative assumption is that hydraulic models have more than a modicum of credibility simply and only because they are physically based.

Coming into the last decade of the twentieth century however, this arguably realist view of river hydraulics was challenged and came to a climatic Dutch finale with the RIVM credibility crisis of 1999 (van der Sluijs 2002). The antidote was the establishment of a framework that includes uncertainty as an integral component.

Good modelling practice

The purpose of a good modelling practice framework is formalisation of the model process in such a way that in following the practice, the modeller has plenty of opportunity to identify and correct mistakes that would otherwise go unnoticed. Of the frameworks proposed we favour the cycle by (Rykiel 1996) as depicted in figure 1. Nonetheless we mention the better known and elaborate ones by RIZA (van Waveren et al. 1999) and recently by Jakeman et al. 2006).

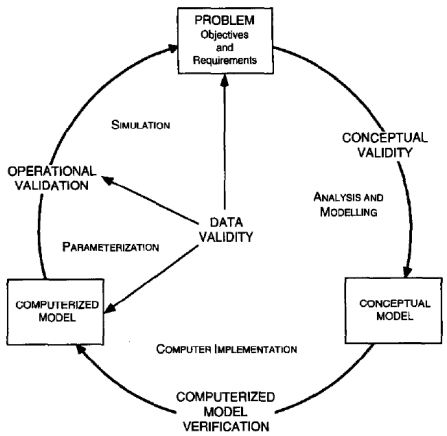

Figure 1 The model process by Rykiel (1996) based on (Sargent 1988) forms the basis of our study

This process description identifies three different stages of validation: conceptual validity (are the shallow water equations appropriate for assessing longitudinal dam construction?), model verification (is the simulated propagation celerity of morphological disturbances in SOBEK 3 consistent with theory?) and operational validation (are the model output variables

consistent with measurements?). Data validity is not a stage, but rather an overarching foundational aspect. The quality of data is so important that it often (deservingly) takes the centre spotlight in uncertainty analysis (garbage in is garbage out).
For those more familiar with the scientific discourse on this topic: we are well familiar with the objections voiced by multiple authors (Konikow & Bredehoeft 1992, Oreskes et al. 1994, Power 1993) on whether or not validation is even remotely possible. While we recognize that traditional validation as proposed by (Klemeš 1986) is on the fast-track to becoming obsolete we agree with Rykiel (1996) that using dictionary synonyms for validation (such as corroboration, confirmation) does not naturally constitutes more clarity. Instead we hope that the meaning of the word will eventually change with the paradigm (as argued by Beck 2002)) and leave it at that.

Assessment of operational validity is an integral aspect to any good-modelling practice and is conditional to how models are used. Within the scope of our study (intervention assessment) we are interested in using models for prediction in out-of-bound situations. The validation requirements for such cases are deceptively simple:

- model output should meet the performance standard
- model applies to more than one case in more than one situation

Both requirements will be discussed briefly.

Validation requirements

The most straightforward validation requirement is in meeting the performance standards. We focus on objective, quantitative measures, although we note that other methods are available such as asking knowledgeable people ('experts') if models are reasonably good at their job. (Warmink et al. 2011) for example, used such methods to quantify uncertainty.
Performance is generally measured by comparing model output to measurements for a specific set of variables. In hydraulic modelling, the water level at a limited set of locations is most often used. Ideally, other variables such as discharges, velocities and salinity are used but those are notoriously difficult to measure directly, which is why in most cases we are stuck with water levels.

To objectively measure the performance of a model, consider the following error model:

$$O(x,t) = M(x,t) - \epsilon(x,t)$$

with (error-free) observation *O*, model output *M* and error ϵ at time *t* and location *x*. Since the error is a function of time and space we use a efficiency metric to summarize the model error. An often used metric is the root mean square error (RMSE):

$$RMSE = \sqrt{\sigma^2 + B^2}$$

with standard deviation σ and bias *B*. In hydrology, the coefficient of determination R^2 (or 'Nash-Sutcliffe efficiency (NSE) criterium' after (Nash & Sutcliffe 1970) is quite popular. The R^2 or *NSE* is generally less useful in hydraulics. This simple assessment of model performance would suffice were it not for calibration. If calibration is necessary – and this is generally the case (Hall 2004) – this validation requirement is significantly more difficult to ascertain and usually determined using the split-sample approach of Klemeš (1986).

The second validation requirement is more difficult to tackle: the model needs to apply to more than one case in more than one situation. This point is better developed by two practical examples.
In more than one situation: in 1995 the river Rhine experienced 'landmark' discharges (although not as high as the 1374 flood (Schielen et al. 2013)) of 12.000 m^3s^{-1}, which is estimated to be a 1 in 50 years event (Chbab 1995). Our safety levels are designed for discharges up to 16.000 m^3s^{-1} as of this moment (the last Room for the River projects are finishing as we speak with the notable exception of Veessen-Wapenveld) and 18.000 m^3s^{-1} in the coming decades. Therefore we need models capable of estimating water levels at 18.000 m^3s^{-1}, while we only have the '95 flood that comes close in terms of validation material; and not ideal validation material regarding the next point.
For more than one case: the river system has changed a bit since 1995. Not only direct anthropogenic change; the Rhine suffers quite a bit from autonomous bed degradation. Our German neighbours suffer the same problem in their realm of the Rhine and kindly warned us of their solution last year (Frings et al. 2014). The whole point of predictive modelling is that the model should perform reasonably well in the new situation compared to the validated situation, but we cannot check that because we still have to find a way to measure the

hypothetical future (in lieu of reliable models).

Validation is furthermore compounded by two other problems: we are – like Theseus' ship – not actually proposing to validate the same model, just one that is similar to some degree. The other problem is calibration.

Obfuscation by calibration?

In calibration, we use part of the available validation data to tweak model parameters (often hydraulic roughness (Warmink et al. 2013)) to improve model performance. Stated differently, we select from all possible other models, viz. the 'same model' with different parameters, only the one that performs optimally during the particular period we have chosen for calibration. Since models are known to behave poorer outside the calibration period (this is known as the 'optimality principle' (Picard & Cook 1984)) we paradoxically do not necessarily select the models that perform well during the intended use. By focussing on a single best parameter set, we might match history, but at the cost that of substantial latent uncertainty (Beck, 2002). To a certain height this can be remedied by modifying the model to better fit the future conditions (Becker et al. 2014), in which case the assumption is made that model parameters can be safely transferred between similar models. In both situations, calibration does not necessarily result in single-best-use parameter settings. Is this really a problem though?

Does it matter?

One might question whether or not problematizing model use in out-of-bound situations really matters:

- intervention effect assessment is assessed relative to reference situations which exhibit the same biases (if there are any)
- in the absence of validation data, the problem is a moot point anyhow
- uncertainty analysis (nowadays a vital part of model validation) is practically infeasible when using complex models

We believe the problem does matter and is worthy of research:

- for research purposes validation data can be synthesized. Also, the Room for the River project coupled with an existing model instrumentarium (Becker et al. 2014) provides interesting case studies.
- Uncertainty analysis is necessary regardless of practical difficulties (cf. (Pappenberger & Beven 2006)). Nevertheless, we will explore more efficient ways of doing uncertainty analysis.
- we (of the greater river studies community) have a responsibility to defend objective credibility of models that transcends practical difficulties. We once again refer to the credibility crisis of 1999 (van der Sluijs, 2002) for framing the societal relevance.

Methodology

We approach the problem in two stages. In the first stage we study the influence of calibration in models of increasing deviation from the reference model in a child-parent set-up (figure 2) and see whether or not continuous recalibration improves model performance.

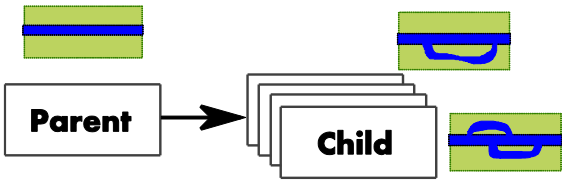

Figure 2 The parent model is calibrated and modified to create child models.

In this stage we take only parameter uncertainty into account for the most important sources of uncertainty as identified by Warmink et al. (2011). For case studies we will use an experimental case with a high dimensional model, possibly extended to a physical scale model. For a real life case we study (part of) the river Rhine using existing measurements and models.

The second stage builds on the first: given information on the parent, how much can we say about the child? We carry out a full Monte Carlo analysis with the parent and use these results to design a much more efficient method for the child using either efficient sampling or an emulator model.

The conclusions from these two stages should enable us to make significant progress towards a feasible methodology for continuous operational validation – including uncertainty analysis – for multi-use models. A possible extension to the model process of Rykiel (1996) is shown in figure 3. This process includes both uncertainty analysis, actualisation and reparameterisation. Part of this process is already practised; we refer to Becker (2014) for a description of actualisation.

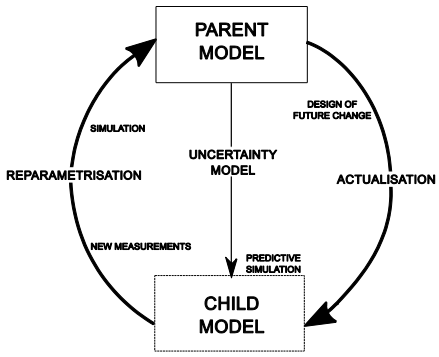

Figure 3 An interpretation of operational validity that takes into account model modification, uncertainty modelling and reparemeterisation

Conclusion: do we have too much confidence?

It might well turn out that our confidence was well placed and that hydraulics and its morphodynamic sibling are spared the trials that plague its nephew, hydrology. There are plenty of signs however that this is not the case (van Vuren 2005, Warmink 2011) and that we do not escape the necessity of uncertainty aware validation of numerical models.

In this paper we have provided several arguments why it is necessary to study not only the environment with our models, but to study our models as well.

As a concluding remark we will now come back to the titular inquiry. The question 'do we have too much confidence in our models' is not the most significant. A more relevant question is: can we defend the credibility of our models? To that question we can only answer that the shifted validation paradigm imposes new challenges that will have to be met. We heartily welcome and hope to help foster discussion on this very important topic.

Acknowledgements

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Holocene lowland stream morphodynamics in a peat-filled valley system

Jasper H.J. Candel[1], Bart Makaske[1], Jakob Wallinga[1], Joep E.A. Storms[2]
[1] *Wageningen University and Research Centre, Soil Geography and Landscape group, P.O. Box 47, 6700AA Wageningen, The Netherlands*
[2] *Delft University of Technology, Faculty of Civil Engineering and Geosciences, P.O. Box 5048, 2628CN Delft, The Netherlands*

Introduction

Stream restoration is an important theme for water managers in the lowlands of Western Europe where low energy streams can be found. Aim is to restore the natural ecological and morphodynamic functioning (Gumiero et al., 2013), and to restore the stream's "natural" state (Newson and Large, 2006). One of the restoration measures commonly taken is the construction of a sinuous channel planform (Kondolf, 2006). Although positive effects have been reported on habitat diversity (Kail et al., 2015; Lorenz et al., 2009), unknown is whether the designed streams are self-sufficient. A self-sufficient stream is defined here as a stream that is able to maintain itself so that no artificial maintenance is needed, which should result in a cost-efficient stream restoration. This could be reached by a stream design that fits the geographical, morphological and hydrological conditions.

A knowledge gap still exists between stream restoration objectives and related practices (e.g. weir removal or re-meandering), and the available knowledge of the morphodynamic functioning of low energy streams (Lespez et al., 2015).
Single-thread, sinuous streams are often seen as "natural" and taken as a reference in stream restoration projects. However, previous studies found evidence that these stream types may be the legacy of historical land use changes which have started around the Bronze Age (Broothaerts et al., 2014; Lespez et al., 2015) or maybe be the result of historical water engineering (e.g. watermills) (Walter and Merritts, 2008). Palaeogeographical investigations on lowland stream evolution during the Holocene will provide insight into morphodynamic functioning of the systems both in their natural and human-influenced states (Grabowski et al., 2014).

In this research, which is part of a four-year PhD research within the STW-funded Dutch RiverCare program, we have focused on lowland streams in a peat-land valley. Lowland streams are often classified as non-dynamic (Eekhout, 2014; Kleinhans and Van den Berg, 2011), especially in peat-covered valleys with relatively highly resistant banks (Micheli and Kirchner, 2002a, b). However, sinuous channels are often found in peat-lands; it is not known whether they are related to lateral migration of channels (Nanson and Cohen, 2014). In addition, it is debated whether such lowland systems should always have a channel, or that drainage may also be dispersed in a wetland system. The latter seems to have been the case during the Middle Holocene in some lowland valleys (Broothaerts et al., 2014; Lespez et al., 2015; Nanson and Cohen, 2014). The objective of this research is to reconstruct the morphodynamic functioning of a lowland stream in a peat-covered valley system during the entire Holocene.

Study Area

The Drentsche Aa catchment (300 km^2) in the northeast Netherlands is a suitable study area since it has been partly peat filled for the entire Holocene (Vos et al., 2011) (Figure). The stream has a length of 28 km (45 km including the side branches) and an mean annual discharge of 1.8 m^3s^{-1}. Stream elevation varies from 16 m +NAP to 0.7 m +NAP. The valleys are located between ice-pushed moraines which are traces of Middle Pleistocene ice sheets. A large part of the area has been covered by coversands during the Weichselian. A large part of the catchment is a national park since 50 years and functions as a nature area. The land use around the national park is forest and agriculture. The Drentsche Aa is one of the few streams in The Netherlands which has never been channelized and still has a sinuous planform (Figure 2).

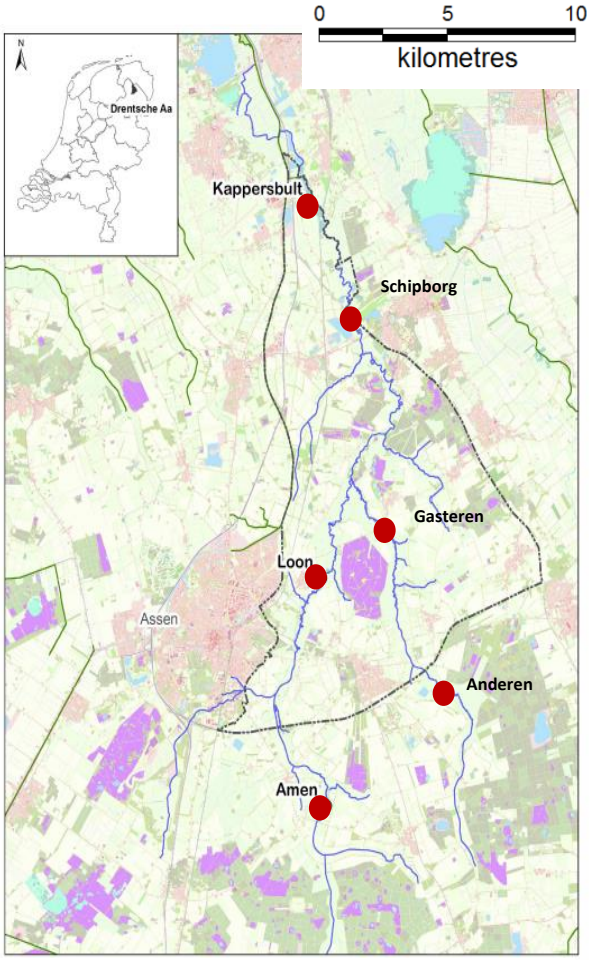

Figure 1 Locations where stratigraphic cross-sections were made. The outlined area represents the National Park. At Gasteren, two stratigraphic cross-sections were made.

Figure 2. Sinuous shape of the Drentsche Aa. Photo taken at Gasteren

Methodology

Seven stratigraphic cross-sections were made perpendicular to the stream with a gouge auger, until the Late Pleistocene coversand was reached. The augering depth varied and was 8 m at maximum. Augering spacing was 5 to 10 m, to ensure that all potential channel lags were sampled. Cross-sections were made in the upper, middle and lower reach of the Drentsche Aa (Figure). A standard method has been followed for the sediment core descriptions (Berendsen and Stouthamer, 2001). Ground Penetrating Radar (GPR) profiles helped us in the stratigraphic interpretation of most cross-sections. Radiocarbon (C_{14}) data by Maas & Makaske (Spek et al., 2015) provides estimates of the peat growth rate at three different locations. Additional C_{14} and Optical Stimulated Luminescence (OSL) dating will be performed for this study.

The preliminary results cover the stratigraphic evolution of the stream channel from the early Holocene until recent. The GPR results cover the top three metres and provide detailed insight into the stratigraphy of the shallow subsurface. Based on preliminary analysis of the data, we infer that a number of mechanisms have contributed to the present sinuous shape of the Drentsche Aa. These will be elaborated upon here and are illustrated in Figure 3:

A) Channel accretion has been entirely vertical. On both sides the channel had and still has peat banks which are very erosion resistant, preventing lateral channel migration. This is the only process which results in a current stream location above the bottom of the Late Pleistocene valley.

B) The channel deposits follow the Late Pleistocene valley during the aggradation of the valley. This is explained by the differences in erosion resistance between the peat and coversand. Peat has a higher erosion resistance than the coversand (Micheli and Kirchner, 2002a, b). The channel is more or less “pulled” to the valley side. However, the erosion rates of the valley wall seem to be relatively low, resulting in a largely unaffected shape of the Pleistocene valley.

C) A combination of the processes in A and B was also found. The channel deposits developed along the axis of the Late Pleistocene valley. When the valley

widened, peat started to grow on the gentle slopes of valley walls causing the channel to accrete vertically.

D) The channel accretion has been vertical (A), until at a certain moment in time an external force results in lateral accretion. This process seems to occur mainly in areas where drift-sands developed. Unknown is how this process works. It might be caused by a channel which regularly gets plugged by drift-sands causing the channel to move laterally to the sides. More research is done to get a better understanding of this process.

B/C+D) It seems likely that the combination of B (or C) and D also occurs, because drift-sands can occur locally. This combination of processes will result in a channel reach located against the Pleistocene valley wall in its current setting.

Because the channel location is influenced by morphology of the valley, the present channel shape is partly a heritage of previous morphology. This may explain the formation of sinuous channels, even in locations with very limited lateral migration. Whether process A instead of process B occurs seems to depend on the slope and width of the Pleistocene valley, because this determines whether peat can grow on the valley sides. The initial stage in the Early Holocene might be crucial for the development of the stream reach during the entire Holocene. No evidence was found of a wetland without a channel. In all stratigraphic cross-sections clear evidence was found of channel deposits at all stratigraphic levels. This finding contrasts with previous ideas of natural streams in similar lowland settings.

Future work

The described concepts will be tested using OSL and C_{14} dating methods, to derive lateral accretion rates, peat growth rates and the periods of drift-sand activity. In addition, the process of drift-sand formation resulting in lateral accretion of streams will be investigated in more detail.

Acknowledgement

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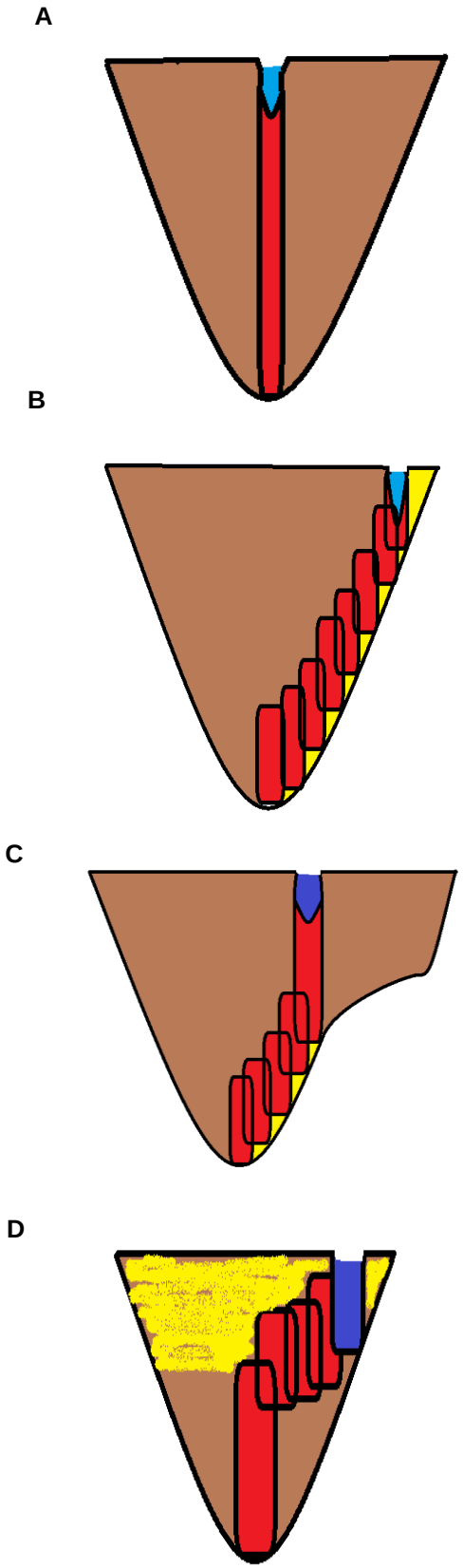

Figure 3. Simplified, schematic cross-sections illustrating different processes that affect the channel planform and morphology in a Holocene valley filled up with peat. Red represents the channeldeposits, brown the peat which mayinclude floodplain deposits, blue the current stream, and yellow the drift sand deposits. A) shows vertically accreted channeldeposit B) shows obliquely accreted channel deposits that are"pushed" to the side by the erosion-resistant peat, C) shows obliquely accreted channel deposits until the channel has lost the connection with the valley side and accrete vertically D) shows how is the process of drift sand activity may affect the style of accretion by causing lateral accretion.

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Limitations when modelling mixed-sediment river morphodynamics

V. Chavarrias[*1], G. Stecca[2,3], A. Siviglia[4], R. J. Labeur[1], and A. Blom[1]
[1]*Faculty of Civil Engineering and Geosciences, Delft University of Technology, Delft, The Netherlands.*
[2]*National Institute of Water and Atmospheric Research, Christchurch, New Zealand.*
[3]*University of Trento, Trento, Italy.*
[4]*ETH Zurich, Zurich, Switzerland.*
[*]*corresponding author; email: v.chavarriasborras@tudelft.nl*

Introduction

Traditionally humans have modified the river system mainly to improve navigability, to reduce flood risk, and to guarantee water extraction for drinking and irrigation. Lately, ecology has received increasingly attention. For designing measures decisions have been traditionally based on empirical knowledge and historical facts. In the last decades, and especially since the availability of computers, mathematical models have been added to the tools of decision-makers.

Although mathematical models have proved to be very powerful tools, they rely on the interpretation of physical phenomena and are therefore limited by our understanding of such phenomena.

Modelling river morhodynamics

For modelling river morphodynamics we apply the conservation principle to the water mass and momentum (i.e. the Saint-Venant (1871) equations), and to the mass of sediment (i.e. the Exner (1920) equation). In case we want to model the behaviour of river sediment of more than one grain size, we apply the conservation principle to each grain size class.

Hirano (1971) was the first to develop a model for mass conservation per grain size. In his model the topmost layer of the bed (the *active layer*) of a specified thickness interacts with the flow. The substrate below only interacts with the active layer. The sediment in the active layer can be entrained and transported, and sediment transported can be deposited in the active layer. If the bed degrades, sediment from the substrate is transferred to the active layer and vice versa if it aggrades. A key point in this model is the fact that the active layer has no vertical stratification, which implies that it is assumed to be fully mixed.

The active layer model has been widely applied, for instance to model downstream fining by selective transport (Hoey and Ferguson, 1994), mountain streams (Sieben, 1997), transport induced by tidal currents and wind waves (Carniello, 2012), gravel-bed rivers with variable discharge (Viparelli, 2011), and it is implemented in commercial morphodynamic models as Telemac (Villaret, 2013), Delft3D (Mosselman, 2008), and BASEMENT (Vetsch, 2006).

However, one drawback of the model is that it may become elliptic, thus being unrepresentative of the physical phenomena we try to model (Ribberink, 1987). In this research we aim to increase our understanding of the problem in order to be able to properly solve it in the future.

Methodology

We first simplify the model seeking to obtain simple analytical solutions to know in which conditions the model becomes elliptic. For that, we assume one-dimensional gradually-varying quasi-steady flow so the Saint-Venant equations reduce to the backwater equation. Furthermore, we assume the sediment mixture to be composed of two size fractions only. With these simplifications, the model is composed of only two equations (plus closure relations) that can be written in a matrix form (Stecca, 2014), as follows:

$$\frac{\partial \mathbf{Q}}{\partial t} + \mathbf{A}\frac{\partial \mathbf{Q}}{\partial x} + \mathbf{B} = 0 \qquad (1)$$

where x and t are the time and space coordinates, $\mathbf{Q}$ is the vector of independent variables, $\mathbf{A}$ is the matrix containing the relations between variables, and $\mathbf{B}$ is the vector containing the source terms:

$$\mathbf{Q} = [\eta, M_{a1}]^T \qquad (2)$$

$$\mathbf{A} = \begin{bmatrix} -\frac{1}{1-Fr^2}\frac{\partial q_b}{\partial h} & \frac{\partial q_b}{\partial M_{a1}} \\ -\frac{1}{1-Fr^2}\left(\frac{\partial q_{b1}}{\partial h} - f_{i1}\frac{\partial q_b}{\partial h}\right) & \frac{\partial q_{b1}}{\partial M_{a1}} - f_{i1}\frac{\partial q_b}{\partial M_{a1}} \end{bmatrix} \qquad (3)$$

$$\mathbf{B} = \left[-\frac{S_f}{1-Fr^2}\frac{\partial q_b}{\partial h}, -\frac{S_f}{1-Fr^2}\left(\frac{\partial q_{b1}}{\partial h} - f_{i1}\frac{\partial q_b}{\partial h}\right)\right]^T \qquad (4)$$

where η is bed elevation, $M_{a1} = F_{a1} L_a$ the mass of fine sediment in the active layer per surface area, F_{a1} the volume fraction of fine sediment in the active layer, L_a the active layer thickness, Fr the Froude number, q_b the sediment transport rate (including pores), q_{b1} the sediment transport rate of the fine fraction, h the water depth, and f_{i1} is the volume fraction of fines at the interface between the active layer and the substrate.

Eq (1) is the advection equation that represents the convection of information over space and time at a finite speed. For a problem governed by the linear wave equation to be well-posed, the speed at which the information propagates needs to be a real number, which implies that the eigenvalues of matrix $\mathbf{A}$ need to be real. Studying matrix $\mathbf{A}$, we find expressions that give us the domain in which the model is not valid.

Analytical analysis

Analysing the characteristic polynomial of matrix $\mathbf{A}$, we find how the behaviour of the system (hyperbolic or elliptic), highly depends on the active layer thickness. The system is always hyperbolic for infinitely large or small values and in between there may be an interval in which it becomes elliptic.

When multiple size fractions are in transport, the size of the neighbour particles affects the transport of a specific fraction. The small grain size classes (relative to a characteristic mean grain size of the mixture) hide behind the large particles. Thus, fine particles experience a larger resistance to be moved by the flow and the opposite happens to the large particles of the mixture. This effect is known as the *hiding effect* and is modelled introducing a hiding factor modifying the critical bed shear stress. Our analysis shows that a large hiding correction leads to an increase of the domain in which the model may lose its hyperbolic character.

The difference between the representative substrate grain size and the one transported is of great importance for the model to become elliptic. Ribberink (1987) and Stecca (2014) show that if the substrate is finer than the transported sediment, model may be elliptic. A large difference between both grain sizes leads to a larger elliptic domain.

Numerical analysis

We check the validity of this analytical analysis assessing the ellipticity of simulations computed with the numerical software suite Delft3D. The reference simulation has a 10 m long 1 m wide domain. The water discharge is 0.140 m^3/s and the upstream sediment discharge is composed of only the coarse fraction and equal to $1 \cdot 10^{-4}$ m^3/s. The fine fraction has a characteristic mean grain size of 2 mm and the coarse fraction of 4 mm. The volume fraction content of the fine fraction in the substrate equals 0.6. The simulation starts in a state of equilibrium and degradation is imposed by a lowering of the downstream water level. The reference simulation is computed using the Meyer-Peter and Müller (1948) transport relation without a hiding correction. This simulation is always hyperbolic (i.e. not elliptic, so representative of a morphodynamic model).

The effect of the active layer thickness in the mathematical behaviour can be appreciated in Fig. 1b and 2b. Fig. 1 shows the stratigraphy at the end of the runs and Fig. 2 the bed elevation profiles over time. The simulation shown in Fig. 1a and 2a (i.e. the reference case, which is always hyperbolic) shows no unexpected behaviour, and is degrading in a continuous way without altering the substrate. On the other hand, in Fig. 1b and 2b, where the active layer is thicker and ellipticity is detected, we see how oscillations appear in the domain that alter the substrate in an unrealistic manner.

Fig. 1c and 2c illustrate of how, by using the same load relation but with the power law hiding function (Parker et al., 1982) the simulation is elliptic. It shows oscillations and an unphysical coarsening of the substrate.

The simulation presented in Fig. 1d and 2d shows how a finer substrate (a volume content of fines equal to 1 rather than 0.6) changes the behaviour of the system.

Finally, we show the effect of the numerical discretization of the domain. The behaviour of the system (whether ellipticity occurs) depends on the equations. However, the consequences of ellipticity, i.e. how the simulation evolves in case it is elliptic, depend on the approach undertaken to solve the equations. When the system becomes elliptic it becomes ill-posed and the consequences are instabilities in the solution that manifest (as previously shown) as oscillations through the domain. The frequency and amplitude of these oscillations increase with the largest wave number, i.e. with smaller horizontal discretization (Hadamart 1922, Joseph and Saut 1990). In Fig. 1e and 2e we show the results of a simulation with the same parameters as in Fig. 1c and 2c yet discretized with a 10 times higher resolution showing oscillations with larger amplitudes.

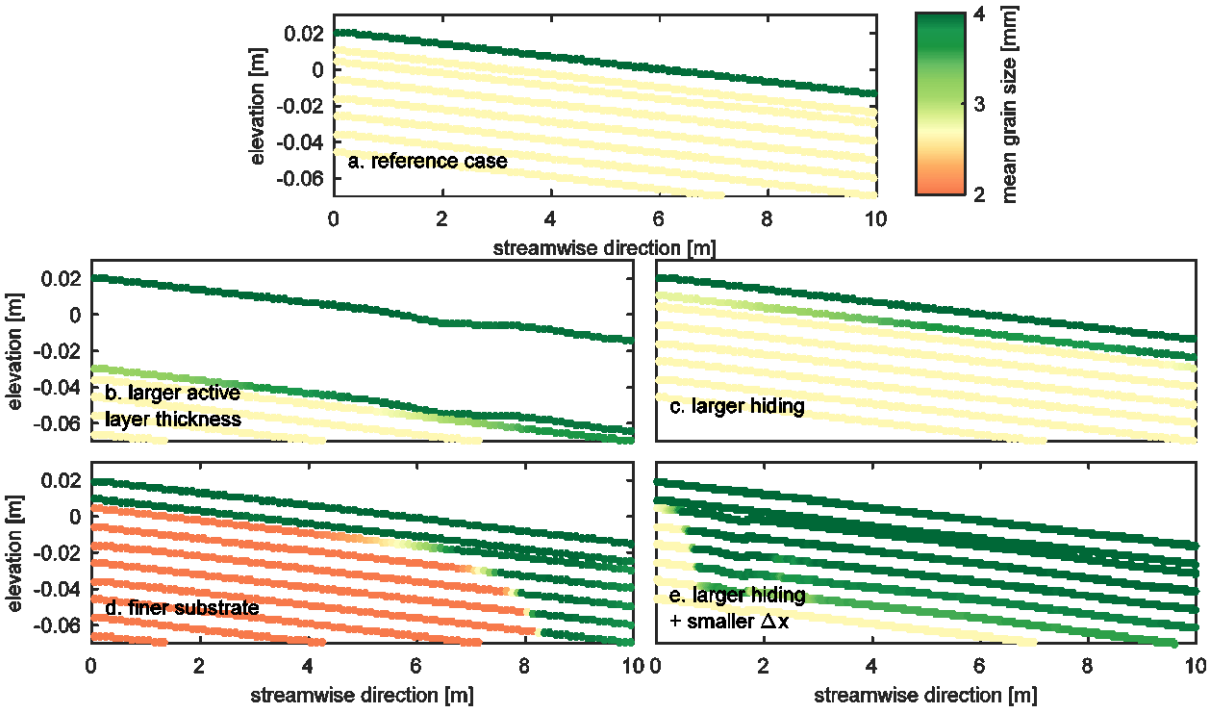

Figure 1. Stratification at the end of the runs (t=10.5h) in (a) the reference case, (b) the reference case with a larger active layer thickness, (c) the reference case with larger hiding correction, (d) the reference case with a finer substrate, and (e) the case with larger hiding correction with a smaller grid size.

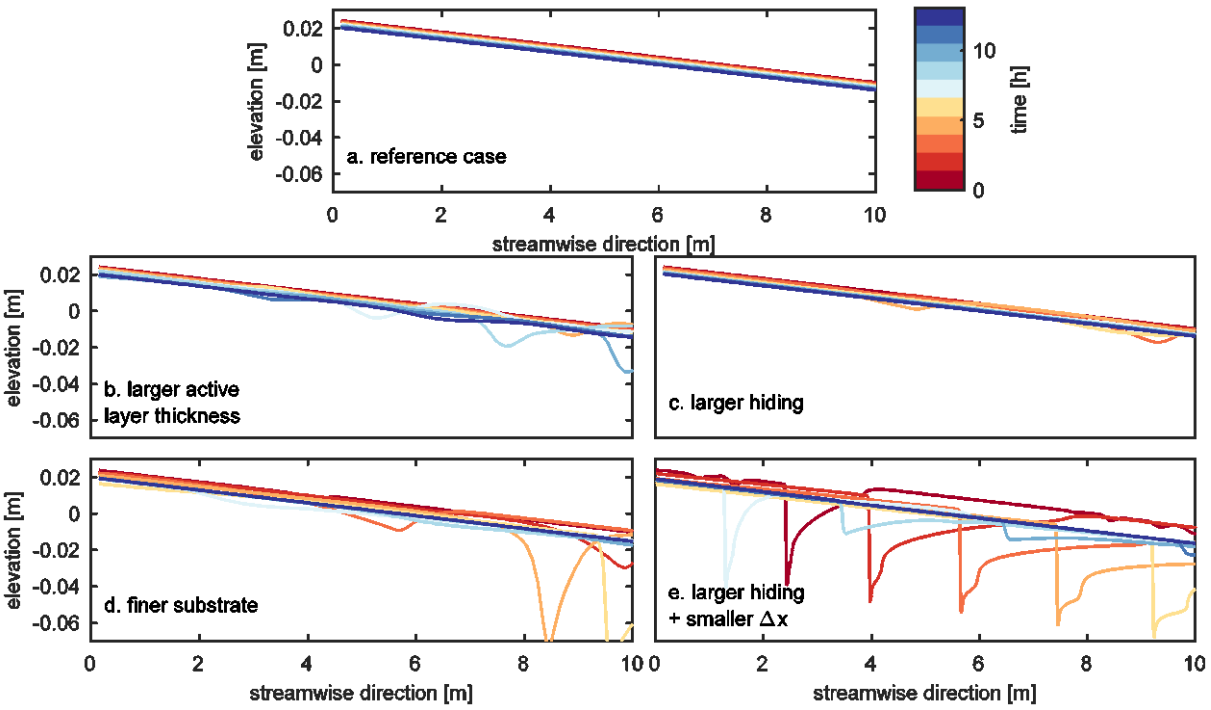

Figure 2. Bed elevation profiles over time in (a) the reference case, (b) the reference case with a larger active layer thickness, (c) the reference case with larger hiding correction, (d) the reference case with a finer substrate, and (e) the case with larger hiding correction with a smaller grid size.

Conclusions

The most common mathematical model used for predicting river morphodynamics with mixed-sediment suffers from ill-posedness. This condition manifests itself as unphysical oscillations that alter the substrate. We conduct an analysis of the system of equations to predict its behaviour (whether elliptic and ill-posed or not). We show the consequences of ellipticity and the role of the main parameters in numerical simulations. We find that:

1. A simulation may be elliptic if the active layer thickness is inside a certain

interval and outside this interval the simulation would be hyperbolic.

2. A hiding correction increases the domain in which a simulation may be elliptic.

We highlight that, as the problem is inherent to the equations, the only solution may come from a better model of conservation of the grain size classes.

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Sensitivity of native and introduced fish species to changes in flow velocity of European rivers

K. Panten[1], F.P.L. Collas[1], A.D. Buijse[2], A.J. Hendriks[1], R.S.E.W. Leuven[1]
[1]Radboud University Nijmegen, Institute for Water and Wetland Research, Department of Environmental Science, P.O. Box 9010, 6500 GL Nijmegen, The Netherlands
[2]DELTARES, Department of Freshwater Ecology and Water Quality, P.O. Box 177, 2600 MH Delft, The Netherlands.

Introduction

In order to evaluate the current and future impact of river modifications on native and non-native fish species assemblages there is a need to collate data on sensitivity of various species groups to individual and multiple environmental stressors (i.e, physical, chemical and biological factors). These data can be used to model and to assess the individual and combined effects of these stressors on native and non-native fish species.

This study focusses on tolerance of riverine fish species to flow velocity because changes in flow velocity conditions are recognized as an important stress factor for fish species. During the last century, most European rivers have been heavily modified to facilitate water use, navigation and other ecosystem services. These modifications alter the physical and hydrological river conditions resulting in changes in river flow velocity (e.g., increase in variability and extreme flow conditions). For instance, rheophilous species will face too low flow velocity in impounded river sections, whereas in free flowing streams an increase in flow velocity variability may cause a limiting factor for limnophilous biodiversity. Navigation induced water displacement adds extreme variation in water flow velocity to already altered flow conditions (Del Signore et al., 2015). In addition to the direct human mediated modifications, climate change affects river flow velocity conditions (Middelkoop et al., 2001; Verzano et al., 2012; Arnell and Gosling, 2013; Van Vliet et al., 2013). Precipitation will become more seasonal and intense, also influencing flow velocity conditions of rivers (Arnell and Gosling, 2013; Van Vliet et al., 2013).

Riverine ecosystems are also colonized by unintentional and deliberately introduced fish species. Differential sensitivity of native and non-native species to changes in flow velocity may affect species composition of riverine ecosystems and competitive interactions between species. We hypothesize that 1) the maximum flow velocity tolerance differ between both species groups since differential sensitivity was also found for several other environmental stressors, such as temperature (Leuven et al., 2011), and 2) that species sensitivity distributions for maximum flow conditions differ for various river catchment in Europe.

Method

The flow velocity sensitivity database constructed by Del Signore et al. (2015) was updated and extended by a literature search using Google Scholar, with the search term consisting of 'Latin species name' and 'flow velocity'. All native and non-native fish species occurring in European rivers were included in this literature search. The first 50 search results were included in the literature review. If a search resulted in less than 50 hits, a second search was performed using only the term 'flow' in combination with the Latin species name.

The flow velocity sensitivities of adult fish species were then analysed using species sensitivity distributions (SSDs), a model that describes the mean sensitivity and the range of sensitivity of a set of species to an environmental limitation (Aldenberg et al., 2002; Posthuma et al., 2002), expressed as the potentially not occurring fraction (PNOF) of species. SSDs were constructed using the statistical software R (R Version 3.2.0; R Core Team, 2015) through fitting a log normal distribution to the data. Subsequently, the mean (μ), standard deviation (StDev, σ) and certainty of each parameter were calculated for each SSDs. Three subsets were made based on the origin of a species: 1) European native species; 2) European species that are present in European rivers outside their native range; 3) Non-European species that were introduced in European rivers.

Differences in PNOF between the three fish species groups were analysed using equation. 1:

$$Z = \frac{x_1 - x_2}{\sqrt{SEx_1 + SEx_2}}$$

where X_1 and X_2 represent either the μ or σ of the compared SSDs and SE_{x1} and SE_{x2} are the standard error of the used

Table 1. The mean tolerance and standard deviation (StDev) of the species sensitivity distributions for the three different subsets using data on maximum flow velocity tolerance of fish species.

	Mean tolerance*	StDev*	Data points (n)
EU native	1.919 (0.045)	0.357 (0.032)	62
EU non-native	1.774 (0.068)	0.347 (0.048)	26
non-EU non-native	1.723 (0.095)	0.454 (0.067)	23

*Standard error between brackets

parameter (Paternoster et al., 1998). With a critical level of 0.05; differences between SSDs were not significant when the *z*-score was between -1.9599 and 1.9599.

In order to test if the PNOF curves varied at different spatial scales, additional comparisons between native and non-native species at a catchment scale were made. The non-native species subset per river consisted of all non-native species irrespective of their origin (European or non-European). Additional data was collected on the fish diversity of five major rivers of Europe. This enabled comparing the PNOF of native and non-native species groups per river. The river Rhine was included in the analyses since it serves as an invasion corridor within Europe. Furthermore, four additional rivers were selected based on their geographical distribution across Europe and data availability (Ebro, Meuse, Vistula and Danube). Subsequently, differences between the derived SSDs were analysed using equation 1.

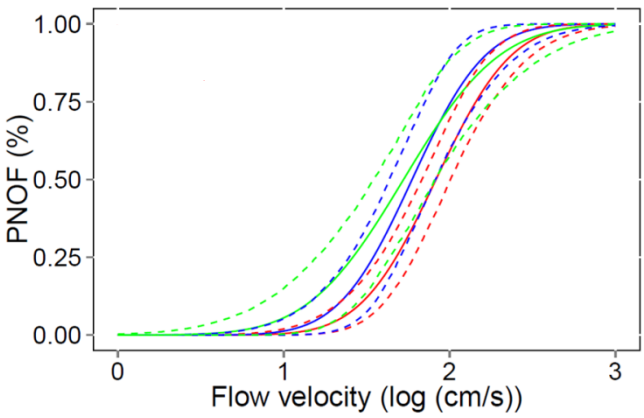

Figure 1. Species sensitivity distributions (SSDs) for log10 transformed maximum flow velocity tolerance of three groups of fish species; Red: European native species; Blue: European species that are present in European rivers outside their native range; Green: Non-European species that were introduced in European rivers. (dotted lines depict the 95% confidence interval; the parameters of each SSD are listed in table 1).

Results

The PNOF curves of native and non-native fish species based on maximum flow velocity sensitivity consistently differ at continental as well as river catchment scale (Fig. 1 and 2). However, the mean and standard deviation of the mean PNOF of native and non-native species groups do not significantly differ (Table 1).

The catchment scale analyses yield similar results as the SSDs at European scale. For all five rivers non-native species assemblages show a slightly higher mean PNOF than native species, but these differences are not statistically significant (Fig. 2; Table 2). Moreover, non-native and native species sensitivity distributions do not significantly differ between the five rivers.

Discussion

No significant difference in mean PNOFs was found between native and non-native adult freshwater fish species on a continental and catchment scale. Thus both hypotheses on differential species sensitivity for maximum flow velocity are not supported by our data. This implies that the SSDs derived at the continental level can be used to predict the PNOF of adult fish species when the maximal flow velocity of a river habitat changes. This is especially useful when data on species composition or species specific tolerance is scarce for a specific river site.

The effect of maximum flow velocity on native and non-native species diversity is equal. However, an increasing number of non-native species are currently dominating fish species assemblages in various riverine habitats. It is possible that other life stages (juveniles, larvae or eggs) of native and non-native species differ in flow velocity sensitivity, thereby explaining recent dominance shifts of species. However, there is not yet enough data available for these life stages to perform a sound SSD-PNOF analysis. In addition, other environmental factors may affect the dominance of non-native species. Therefore, several other environmental factors should be included in effect predictions of future flow velocity conditions on the establishment of non-native fish species.

An effort should be made to perform research on the flow velocity tolerance of different life stages of both native and non-native fish species. Furthermore, data on

other stress factors should be collated in order to identify other limitations of native fish species that might explain a dominance shift to non-native species (e.g. temperature).

Conclusions

- SSDs for maximum flow velocity tolerance of native and non-native adult fish species did not differ significantly at continental as well as river catchment scale.
- The derived SSDs can be used to predict the potential effects of changes in maximum flow velocity on PNOF of adult fish species at various spatial scales (i.e., location, river and continent).
- In order to support appropriate ecological impact assessments and biodiversity modelling we recommend to derive SSDs for 1) minimum and optimum flow velocity, 2) variability in flow conditions, 3) different life stages, guilds and migratory groups, 4) subsets of non-native species (deliberately introduced versus unintentional), 5) other environmental factors, and 6) their combined effects.

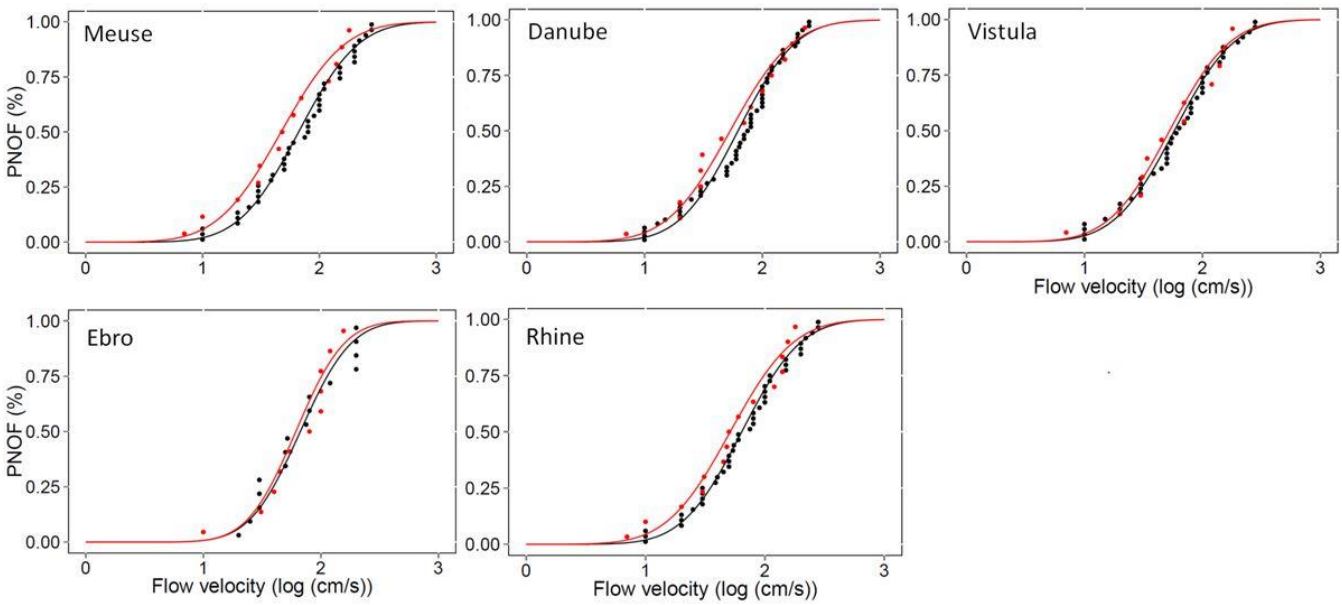

Figure 2. Species sensitivity distributions (SSDs) for log10 transformed maximum flow velocity tolerance of native (black) and non-native (red) species in the five European rivers.

Table 2. The mean tolerance and standard deviation (StDev) of the species sensitivity distributions for the five European rivers using data on maximum flow velocity tolerance of fish species.

River	Status	Mean tolerance*	StDev*	Data points (n)
Ebro	non-native	1.788 (0.098)	0.325 (0.069)	11
	native	1.826 (0.085)	0.340 (0.060)	13
Meuse	non-native	1.673 (0.119)	0.429 (0.084)	12
	native	1.823 (0.063)	0.401 (0.044)	10
Vistula	non-native	1.722 (0.116)	0.403 (0.082)	15
	native	1.763 (0.060)	0.398 (0.042)	14
Rhine	non-native	1.710 (0.108)	0.418 (0.076)	44
	native	1.796 (0.060)	0.390 (0.043)	43
Danube	non-native	1.727 (0.112)	0.420 (0.079)	42
	native	1.794 (0.052)	0.387 (0.037)	55

*Standard error between brackets

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The potential of web-collaborative platforms to support knowledge exchange in river management

V.J. Cortes Arevalo[1], S.J.M.H. Hulscher[1] & M.C. van der Voort[2]
[1]Department of Water Engineering and Management, [2]Human Centred Design at the Faculty of Engineering Technology, University of Twente, PO Box 217, 7500 AE Enschede, The Netherlands

Knowledge exchange in river management

Adaptive and integrated river management is increasingly required by European directives to cope with changing river systems and multi-functional requirements (Pahl-Wostl 2006). Such perspective becomes evident in policy guidance for flood protection (European Commission 2007) and river restoration prjects through national river programmes such as "Room for the river" in the Netherlands. Moreover, the importance of stakeholder involvement in river management and related decision-making processes has been widely recognized (Reed 2008).

Borrowing the definition of Baede et al. (2007) from Aye et al (2015), stakeholder is related to the variety of scientific, institutional and local organizations that have a legitimate interest or may be affected by river management actions. Information sharing and knowledge exchange between relevant stakeholders can lead to more collaborative interactions, for example in decision-making (Failing, Gregory, and Harstone 2007; Edelenbos, van Buuren, and van Schie 2011). In river management, the collaboration between different stakeholder groups into a multi-disciplinary environment differ according to the level of stakeholder participation and governance style (Neuvel and Van Der knaap 2010). Knowledge exchange can be for example, in the form of guidance for project formulation, understanding and interpreting available information, communicating uncertainties and underlying assumptions, identifying critical points, elicitation of decision-making criteria, evaluating and negotiating about management options (e.g. Evers et al. 2012; Voinov and Bousquet 2010).

Stakeholder participation traditionally ranges from information sharing or one-way communication to more interactive or two-way communication forms (Arnstein 1969). However, Wehn et al. (2015) highlights the importance of contextual aspects of participation as referred by Fung (2006). Thereby, the goals of involvement, those who actually participate and the ways in which they are invited to do so, become relevant to support collaborative processes and knowledge exchange. However, the extent to which collaborative interactions are achieved and translated into meaningful knowledge for river management is still under debate (Wehn and Evers 2015).

Therefore, a preliminary but important pre-requisite to support collaborative interactions is facilitating a knowledge management base through information (ISs) and decision support systems (DSSs). Web-collaborative platforms offer important opportunities to combine capabilities of such systems for information sharing and supporting tools for river management. Moreover, guiding principles for collaborative interactions (Reed et al. 2014) should be adapted according to the specific study area and river management phase. Management phases often comprise a decision-making process for problem identification and framing, formulation of management alternatives, alternatives evaluation and negotiation, monitoring and re-evaluation.

User-centred design of web-based information and support systems

Advances in web-based GIS and information technologies have increasingly supported the development of ISs and DSSs. Standardization of technologies and open-source innovations have facilitated management of geo-spatial data, information sharing as well as visualization and analysis of relevant information for river management (Choi, Engel, and Farnsworth 2005).

Such systems usually comprise of a three-tier architecture for the data model (including metadata), logic and presentation functionalities (See Figure 1). In river management, ISs should generally satisfy a multi-stakeholder environment that often includes non-specialists. Therefore, user groups have different skills, information needs, and various degrees of data synthesis and analysis requirements (Mcdonnell 2008).

Moreover, DSSs go beyond IFs by assisting interpretation of available knowledge for defining decision criteria, formulating and evaluating management options through a variety of methods, for example, multi-criteria evaluation (Matthies, Giupponi, and Ostendorf 2007). When collaborative interactions exist, access to available data, information and knowledge is provided to participant stakeholders (user groups) according to accessibility rights. Different knowledge types as well as related uncertainties (e.g. lack of information) must be clearly communicated (e.g. metadata) to comprehensively frame river management problems. Scientific knowledge comprise available data from official sources and modelling outputs. Direct interaction with scientific models into the system is initially not considered to limit complexity to the variety of user groups. Instead, systems may incorporate interactive elements such as multimedia resources, management of comments and role playing (see Fig. 1). The specific case is under consideration.

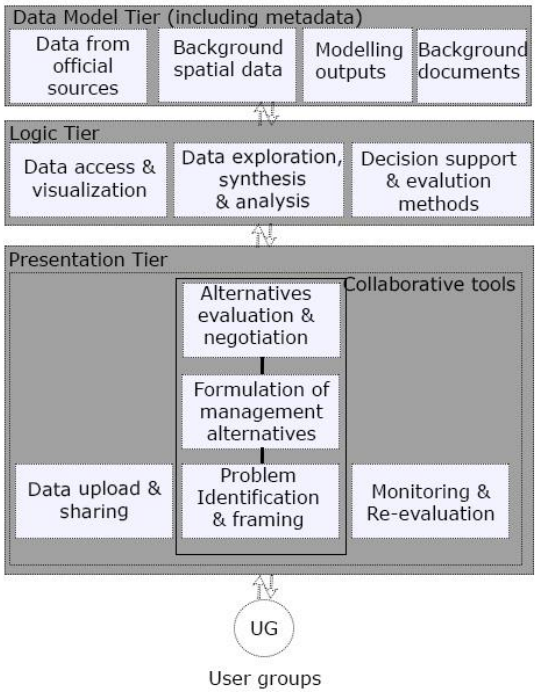

Figure 1. Components of a knowledge management base. (adapted from Gouveia et al. 2004)

In order to understand, categorize and prioritize user groups' requirements, we recognize the importance of user-centred design approaches since early development stages (Bijl-Brouwer and Voort 2008; Cortes Arevalo et al. Submitted). Moreover, relevant research questions (RQ) for the design of such systems include (Mcdonnell 2008; McIntosh et al. 2011):

- RQ1: Who and what are the roles and needs of data/knowledge providers and users?
- RQ2: What are the core-functionalities for the different user groups to support data sharing and knowledge exchange for decision support?
- RQ3: What are the components, workflow and data management model for the different user cases and which collaborative are required?
- RQ4: Which system architecture is the most appropriated according to available funding and technological resources towards reusing, upgrading and maintaining the system in long-term basis?
- RQ5: How can we test and validate usefulness and applicability criteria according to the context of use? For example a river management phase or decision-making process in an specific study area.

Such questions are generally part of a software development cycle of ISs and DSSs (Gulliksen et al. 2003). Thereby, prototypes are developed and refined incrementally (Fig. 2). Focus of this research is on designing a web-platform prototype with core-functionalities supporting usefulness.

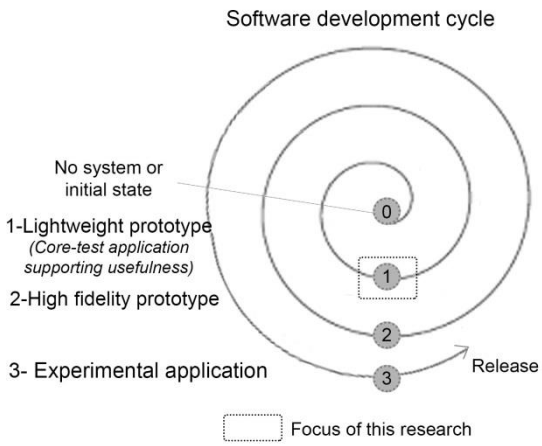

Figure 2. Software development cycle based on Mysiak, Giupponi, and Rosato (2005)

Potential usefulness and applicability: RiverCare Project context

River restoration projects have been implemented in many developed countries to address multi-functional requirements of river systems such as safety, navigation and biodiversity (e.g. ECRR 2015; REFORM 2015). However, stakeholders should better understand the long-term effects of river interventions in dynamic natural river processes such as erosion and sedimentation. That knowledge is particularly important, not

only to reduce maintenance costs, but also to increase awareness about perceived benefits of river interventions (Schielen, Augustijn, and Hulscher 2015).

Thereby, we aim at evaluating the potential usefulness and applicability of web-collaborative platforms for information sharing and knowledge exchange for decision support in river management. We understand usefulness criteria according to the extent in which users believe that a system is useful to perform their activities and it is appropriate for the context of use (Laitenberger and Dreyer 1998). Although usefulness is a preliminary requirement for usability, it is important to define the functionalities of the system (Bevan 1999).

In the context of this research, knowledge providers comprise but are not limited to the RiverCare researchers. Knowledge users are initially represented by the institutes, companies and governmental bodies in the Netherlands that integrate the user staff committees. Methods will be implemented in coordination with researchers of RiverCare that share user requirements. Important consideration is given to support knowledge exchange by means of collaborative tools. Both collaborative tools and usefulness criteria will be the user requirements output.

Design steps account for user requirements, design and implementation, testing and evaluation of the first development stage of a web-platform prototype. Therefore, the user requirement analysis stage will account for different methods such as interviews, questionnaires, focus groups and participatory design workshops to define both user needs and usefulness criteria. For the design and implementation in Table 1, modular components of applications such as OpenEarth are under consideration following implemented web-platforms examples (Van Koningsveld et al. 2013). For evaluation and testing, one of the pilot cases of the project will be considered.

Concluding remarks

Although the intended *outcome* of this research is the design of a web-collaborative platform. Such *outcome* will not automatically imply a higher stakeholder involvement and knowledge exchange for river management (Wehn and Evers 2015). Thus, from the scientific perspective, the research aims at evaluating the usefulness and applicability of such *outcome* as a *mean* towards collaborative interactions. Gaps between design and use require to start simple, to combine ease of use, usefulness and validity of methods implemented. Moreover, the design process requires to agree on accountable outputs with participant stakeholders according to available funding and resources for development (Junier and Mostert 2014; McIntosh et al. 2008).

Aknowledgements

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Analyzing public perceptions of flood safety in relation to riverine vegetation in floodplains: a photo-questionnaire

J.A.N. Dijkwel[1], J.M. Fliervoet[1], L.N.H. Verbrugge[1]
[1]Radboud University Nijmegen, Institute for Science, Innovation and Society, P.O. Box 9010, 6500 GL Nijmegen, The Netherlands

Introduction

The planning and implementation phase of the Room for the River program will be finished at the end of this year. Therefore the river management in the Netherlands is shifting towards maintaining multi-functional floodplains. The management of flood safety and riverine ecology is the core aspect for this perception-research.

Nowadays, maintenance measures have to deal with flood safety aspects and riverine ecology in a harmonized way. One of the measures used in floodplain management is the strategy of Cyclic Floodplain Rejuvenation (CFR), what can be seen as a possible solution to prevent major flooding. This strategy imitates the effects of channel migration by removal of soft wood forests, by lowering floodplains and by (re)constructing secondary channels. Hydraulic roughness will be reduced and the cross-sectional area of the floodplain will be larger again (Baptist et al., 2004).

In the past, most floodplain land-use practice prevented natural vegetation succession (Baptist et al., 2004). This is changed during the past years and is still changing towards more natural floodplains. Previous studies focused on vegetation resistance in floodplains per vegetation type and how the vegetation is structured in the area. This research will contribute to this body of knowledge by studying public perceptions towards aesthetics, naturalness, flood safety and need for management measures in relation to the riparian vegetation in floodplains. Therefore, the following research question and sub-questions were formulated: *What are public perceptions of aesthetics, naturalness, flood safety and the need for management measures in relation to riparian vegetation in floodplains?*

1. Do different vegetation types in floodplains trigger a different perception of safety?
2. Does dispersion of vegetation have an influence on the perception of flood safety management?
3. What aesthetic values do certain vegetation features have concerning flood safety in the eyes of the respondents?
4. Which vegetation types have the preference when looking at the naturalness of floodplains?

This article explores the perceptions of the respondents using a photo-questionnaire and discusses how this is related to other research on public perceptions in water management (cf. Le Lay et al., 2008).

Method

In this pilot study we used photographs of floodplains to explore public perceptions of aesthetics, naturalness, flood safety and need for management measures. The criteria for selecting photographs were based on: vegetation types like trees, thicket and grass, and structures, like dispersion and hedges..

The questionnaire used five questions that had to be filled in by the respondents for the ten photographs in the questionnaire (on the four aspects, including an additional management question). The respondents were students from the Transnational eco-system based Water Management Master track (TWM). Paired t-tests were performed to check the differences in answers for the different categories between the photos. The aspects that were assessed, analyzed and evaluated are the following:

1. Aesthetics (how aesthetically pleasing is the floodplain)
2. Naturalness (how natural is the floodplain)
3. Flood safety (how beneficial in coping with floods is the floodplain)
4. Need for management measures (if there is need for improvement)

The photographs that were used in this research are presented below (Figure 1):

Figure 1: Floodplain photographs 1-10, used in the research.

Background of the respondents

Before answering the questions of the aspects, demographics data were collected. The respondents (N=16) had an average age of 25 years, ranging from the age of 23 to the age of 29. Further, the group contained 3 female and 13 male persons. The majority of the students came from Germany (N=7). The Bachelor studies that the students completed prior to the TWM track are in the science field; the majority of the respondents studied Biology (N=5). The majority of the students grew up in a city (79%) and the others in a rural area (21%).

Results

Average values per photo concerning the four aspects are shown in Figure 2.

Scenic beauty was scored lowest for the photos containing grass (grass1 and grass2) and the photo where a hedge is present (hedges2). Highest average values were given to the photos with the dispersed vegetation structures (dispersion1 and dispersion2), but also to the fourth photo (thicket1).

Figure 2b shows high values for naturalness for the photos containing dispersed vegetation both grouped and

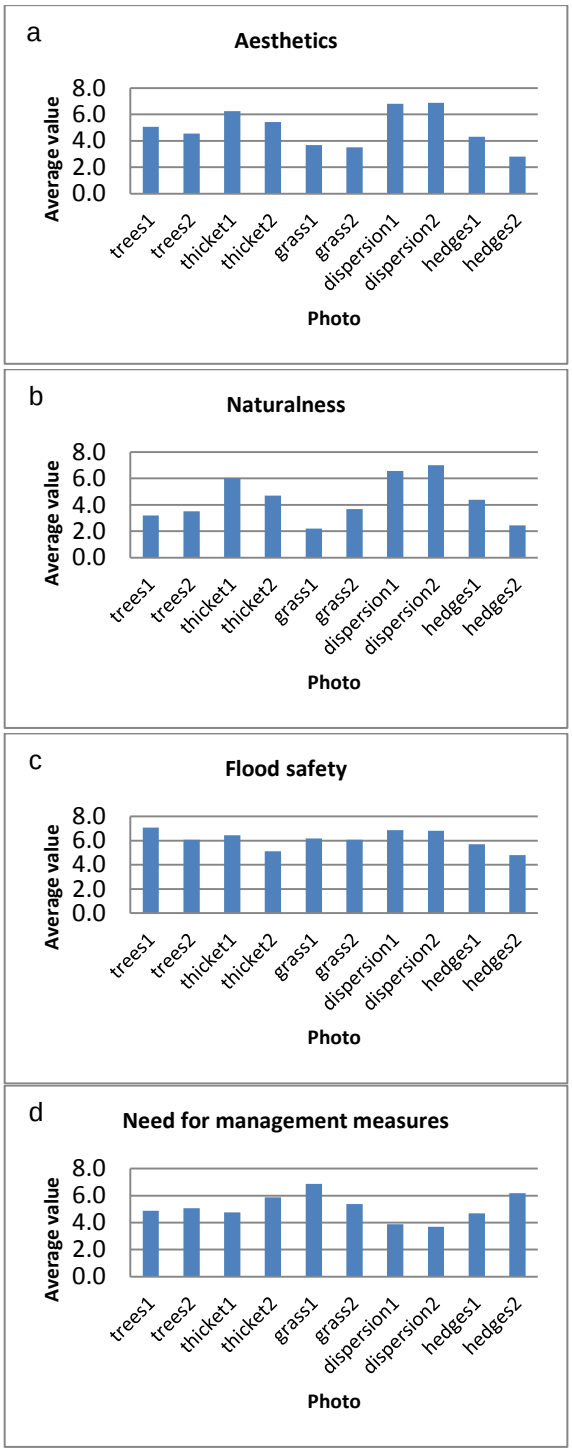

Figure 2: Average scores of the photos for, (a) aesthetics, (b) naturalness, (c) flood safety, (d) need for management measures.

single random vegetation (dispersion1 and dispersion2) and for the photograph with thicket far from the river (thicket1). Lower values for naturalness were given for the photograph with trees near the river (trees1), both photographs with grass (grass1 and grass2) and photograph 9 with hedges longitudinal to the river.

Figure 2c, about flood safety, shows that floodplains are perceived to be the least

beneficial in coping with floods were depicted in photos 6 with thicket near the river (thicket2) and 9 with hedges longitudinal to the river (hedges2). The photos with trees near the river (trees1) and dispersed vegetation (dispersion1 and dispersion2) were the floodplain situations that can were perceived to cope with floods the best.

Figure 2d, shows that the floodplains with cut grass (grass1), thicket near the river (thicket2) and hedges longitudinal to the river (hedges2) are judged to be the most in need of management measures. The photos with dispersed vegetation have the lowest values and thus management measures are least needed.

Management measures

The ten photographs were ranked according to their average scores for need for management measures. Scenic beauty, naturalness and flood safety were the management measures the respondents could choose from (Table 1). For the three aspects of management measures (i.e. naturalness, scenic beauty and flood safety), all were marked at least once.

Table 1: Average values per photo on the need for management measures.

Photo number	Vegetation types-structures	Average value	Most marked management measure	Total N
3	Grass1	6.9	Naturalness (n=9)	14
9	Hedges2	6.2	Naturalness (n=12)	14
6	Thicket2	5.9	Naturalness (n=7)	15
5	Grass2	5.4	Scenic beauty (n=7)	14
10	Trees2	5.1	Naturalness (n=8)	13
1	Trees1	4.9	Naturalness (n=10)	14
4	Thicket1	4.8	Flood safety (n=5)	12
7	Hedges1	4.7	Naturalness (n=4)	12
2	Dispersion1	3.9	Flood safety (n=6)	13
8	Dispersion2	3.7	Scenic beauty (n=4), Flood safety (n=4)	10

Discussion

Results of the questionnaire indicate that there were different public perceptions triggered by the different vegetation types and structures used in this research. Only for a few photographs, the flood safety level is low (around the value of 5) (Figure 2c) and the respondents wanted to see this managed (Table 1). It showed that the vegetation types had influence on the public perceptions, because when a photograph had grass as main aspect, the respondents did not see a real danger towards flood safety in contrast to the photographs that contained trees and thicket. An important trend is when respondents see more grass in the photograph, the beneficialness of the floodplain to cope with floods increases as the floodplain has room for processing high water levels. When more trees are present in the photo, the floodplain gets less beneficial according to the respondents. This corresponds to the low hydrologic resistance of grasslands.

Further, this study was also conducted to see if the perceptions of dispersed vegetation differ from the other vegetation situations like hedges or grass filled floodplains. Even though the two floodplains photos that both show dispersed vegetation indicate that it does have influence on the perception on flood safety, the difference between single (dispersion1) and grouped (dispersion2) vegetation did not really matter according to the respondents (see Figure 2a-d). Vegetation in the streamflow of the floodplains, especially thicket, has the biggest resistance to water (Willems, 2013). The photographs with dispersed vegetation also contain thicket, so we would expect that floodplains with thicket and trees, would be less beneficial, but this was not found in this public perception study. These floodplains show the most beneficial values for coping with floods (Figure 2c). This raises a dilemma, because the public perception is in contrast to the situation that has the most benefit for flood safety proven by previous research (Willems, 2013). In all probability, the respondents did not know this fact and thus did not notice that these floodplains with random dispersed vegetation have a negative influence on the flood safety in floodplains, even though the respondents have already some knowledge on water management. Public perception could be changed by creating more awareness by the water managers and so, creating more understanding of floodplain issues by the public.

Focusing on the aspects of aesthetics and naturalness, the same pattern is clearly present throughout the photo 1 till 10 (Figure 2a and 2b). The values for naturalness fluctuated more compared to the values of aesthetics. Floodplains with dispersed vegetation show the highest values on aesthetics and score quite good (values > 6.5) on naturalness. When looking at the photographs with hedges, the floodplain with a perpendicular hedge to the river scored higher for flood safety (more beneficial) than a floodplain with a longitudinal hedge. This is in contrast to previous studies on the damming of water in areas near the river (Staatsbosbeheer, 2013). Remarkably, even though hedges (rows of trees and thicket) often have a high

cultural historic value connected with a high aesthetics value (Staatsbosbeheer, 2013), this was not found in this study.

Implications

The quality of this pilot-study will benefit from collecting more data. This could be realized by surveying a larger group of respondents, including different target groups. Furthermore, a point for improvement of this study is that this research did not take the spaciousness into account and this should be considered in other studies on this subject. Spaciousness is an aspect that is used frequently in measures for the Room for the River program. For example, a dike replacement further away from the river containing lands with grass creates more space for coping with flood water (Van Vuren et al., 2005). Furthermore, another study concluded that flood water could be controlled with the creation of retention areas, behind high roughness areas near the river caused by riparian vegetation. And forms an alternative solution to flooding (Greco & Larsen 2014). Thus, taking the aspect of spaciousness into account, as well as a larger dataset (e.g. different target groups), would contribute to our understanding and create more depth to a follow-up research.

Conclusion

We conclude that the preferred floodplains concerning the aspects of aesthetics, naturalness and flood safety are floodplains that present random dispersed vegetation and thicket close and far from the river.

Aesthetics and naturalness had the same tendency throughout photograph number 1 to number 10 in the average scores given. For aesthetics and naturalness, the floodplains that showed random dispersed vegetation, showed higher average values. For naturalness the peaks in the highest score differ the most with the mean indicating that the aspect of naturalness has the largest range. The floodplains with grass and hedges, show the lowest average values for aesthetics and naturalness. In naturalness, for these lower classed floodplains, wild grass has preference above cut grass. Additionally, a hedge that is perpendicular to the river had its preference above a hedge that is longitudinal to the river. A trend is observed in the aspect of naturalness, higher scores are given when there are more trees and thicket in the photograph, and when the amount of grass increases in the photo the aspect of naturalness declines.

Furthermore, when there are more trees visible in the floodplain, the beneficialness declines in coping with flooding. When more grass is present in the photo, the main perception was that these floodplains deal better with floods than trees and thicket.

In addition, we was asked what kind of management measures were needed according to the respondents. This led to the fact that most floodplains had to be changed in degree of naturalness. The the floodplains with dispersed vegetation had the lowest scores for need of management measures, even though these are the ones that will cause more hydraulic resistance than the other floodplains.

Acknowledgements

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Research on Natural Bank Erosion Processes

Gonzalo Duró*[1], Alessandra Crosato[1,2], W.S.J. Uijttewaal[1]
[1] *Delft University of Technology, Section of Environmental Fluid Mechanics, Faculty of Civil Engineering and Geosciences, PO Box 5048, 2600 GA Delft, the Netherlands*
[2] *UNESCO-IHE, Department of Water Engineering, PO Box 3015, 2601 DA Delft, the Netherlands*
** Corresponding author; e-mail: g.duro@tudelft.nl*

Introduction

Rivers are part of the natural environment. Within their many patterns and characteristics, bank erosion is widespread in natural conditions and may lead to channel width adjustments (Thorne 1998). Also, bank erosion as found in nature is a process that allows for the presence of diverse habitats for aquatic plants and animals, which enhances the river ecological functions (Florsheim et al. 2008). However, the reduction or absence of bank erosion is often sought in favour of diverse human needs, such as navigation and flood protection, or simply to avoid property damage and land loss. It is thus a challenge to achieve a river training or restoration strategy that integrates human development and ecological functions.

The importance of accurately determining the conditions and rate of bank erosion involves both large-scale landscape processes (e.g. channel patterns and channel migration) and reach to local scales where engineering projects regularly dwell.

Figure 1. Naturally eroding bank at a restored branch of the Vecht River. The restoration project allows for channel migration.

Objectives

The main goals of this project are to deepen the understanding of bank erosion processes and to explore more natural measures to manage them compared to traditional ones (e.g. groynes). A proper understanding and predictability of the involved morphodynamics are then key in the design of river restoration projects. This study is carried out as part of the project 'RiverCare' and it is at an initial stage of development, thus the strategies for achieving these goals are still in progress.

Bank erosion

The erosion process of a riverbank represents a geometrical change of a river cross-section, which extends along a certain length. Figure 2 shows a conceptual framework in the spirit of analysing the overall process with its different mechanisms, characteristics of the scene and the internal and external acting forces.

There are several mechanisms that promote this to happen, but two essential ones are the progressive direct entrainment of grains that constitute the bank surface and the mass waste by geotechnical failure. The former mode may occur under fluvial or wave erosion that removes cohesive and/or non-cohesive sediments, in the first case driven by the action of hydrodynamic forces, or by wind or vessels in the second case. As a remark, vessels can also influence the near-bank hydrodynamics, thus influencing fluvial erosion as well. On the other hand, banks may fail because of geotechnical instability. This can happen by toe erosion (fluvial erosion) and/or bank undermining due to piping, being both mechanisms affected by the bank characteristics. Bank resistance is usually characterized by an angle of internal friction and a cohesion parameter (ϕ and C), but they may present different layers of soil compositions adding complexity to the analysis. What is more, the presence of riparian vegetation and pore water also affects the bank resistance against failure, which both depend on climate conditions. Finally, subaerial processes (e.g. weathering) act as an erosion agent and influences the soil strength.

Methods

The first research goal will be addressed by performing laboratory tests at a reduced scale with the aim of deepening the understanding of the bend morphodynamics, including bank erosion. Curved flows are abundant in natural conditions and the detailed hydro- and fluvial

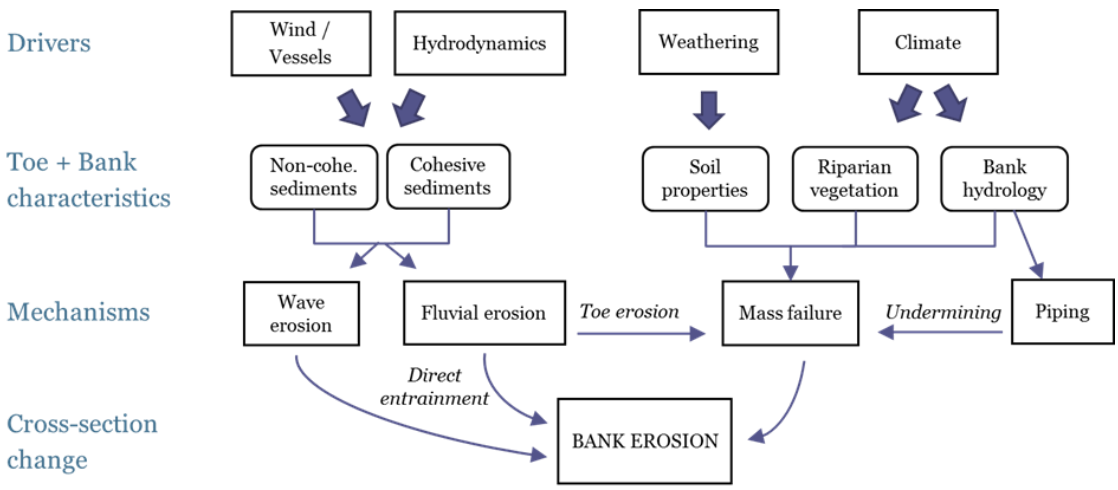

Figure 2. Conceptual framework of bank erosion. Primary drivers of bank retreat applied to a complex system of inert particles, biomass and water result in different erosion mechanisms.

dynamics, despite important advances in recent years (Koken et al., 2013), still lacks of understanding regarding the bank erosion process and lateral migration.

Complementary to laboratory tests, a CFD code may serve to analyse the flow structures (Figure 3) and compare/contrast results. In addition, the CFD implementation can help interpreting scale effects, which can be important regarding secondary flows (Koken et al., 2013).

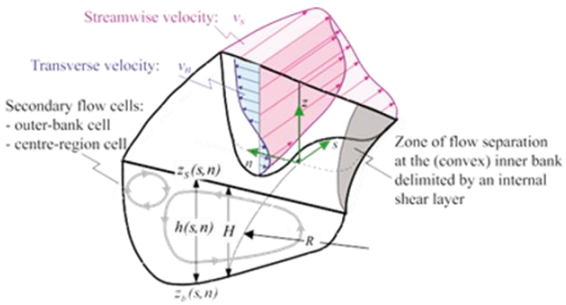

Figure 3. Conceptual representation of flow features at channel bends (from Blanckaert et al. 2012).

Finally, field surveys and tests will be carried out, given the complexity that vegetation introduces in bank erosion studies (e.g. Hubble et al., 2010).

It is of particular interest to this research to explore the possibilities of using riparian vegetation to stabilize riverbanks. Previous experiences (e.g. Abernethy and Rutherfurd, 1999) will serve as benchmarks to adapt methodologies to local environments.

Near-future work

Further research will focus on identifying the most promising set-up for laboratory experiments to investigate bend morpho-dynamics, selecting relevant field sites to monitor natural erosion and planning field campaigns.

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Probability of simultaneous occurrence of discharge peaks in the main branch of the Meuse River and its tributaries

T.J. Geertsema[1], A.J.F. Hoitink[1], A.J. Teuling[1], P.J.J.F. Torfs[1], A.H. Weerts[2] and R. Uijlenhoet[1]

[1] Wageningen University, Hydrology and Quantitative Water Management Group, P.O. Box 47, 6708 AP Wageningen, the Netherlands

[2] Deltares, P.O. Box 177, 2600 MH Delft, the Netherlands

** Corresponding author; e-mail: tjitske.geertsema@wur.nl*

Introduction

The current water policy aims at accelerating discharge to mitigate floods, which are causing problems, such as eutrophication, reduced (aquatic) biodiversity (Bernhardt & Palmer 2011), drastic decline in groundwater levels (Van Asselen et al., 2010; Hatala et al., 2012; Brown and Nicholls 2015), increased water levels during floods and prolonged drought, predominantly in urban areas and densely populated lowland areas. To address and prevent these problems, the water policy should concentrate more on water retention according to STOWA, Dutch Foundation of Applied Water Research. Few of the existing strategies towards water retention have been evaluated regarding their functionality and quantified impacts on the regional water system.

Delaying discharge might provide a local solution for water retention, but could lead to discharge peaks that regionally coincide, with major consequences of flooding. The purpose of this study is to quantify the probability of simultaneous occurrence. It is known that floods naturally coincide in the Walloon Ardennes of the Meuse near Liège (Fig. 1 and De Wit et al., 2007). To understand the characteristics and effects of simultaneous occurrence, the whole Meuse catchment including the Dutch streams will function as research area. The probability will be defined in time and on the basis of conditions by catchment characteristics and precipitation. Also, the effects of simultaneous occurrence on the tributaries are studied. The following research questions are defined:

- Under which circumstances do discharge peaks coincide?
- Which characteristics of the catchments show a relation with the simultaneous occurrence of discharge peaks?
- What are the quantified water level effects of the simultaneous occurrence of discharge peaks?

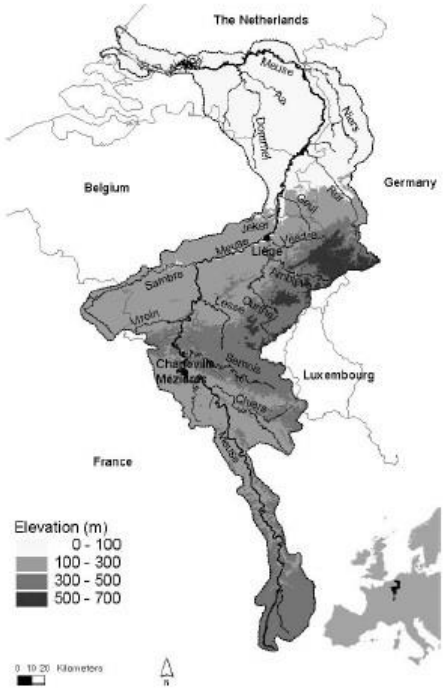

Figure 1. Location of the Meuse basin (derived by De Wit et al. 2007).

Methodology

The probability of simultaneous occurrence of discharge peaks will be determined by the probability that there is no time shift of the peaks in the Meuse River and in its tributaries. The time shift of the discharge peaks will be determined using dynamic time warping (DTW). DTW is a unique peak-time analysis in considering the time elastic / dynamic in the X-axis. This is desirable for similar shapes with different phases, like long wave propagation. Moreover, DTW is not computational expansive; therefore large time series can be used. DTW has been introduced by Berndt and Clifford (1994) and a detailed description is written by Keogh (2002).

This method finds the smallest cumulative difference between two time series (Ouyang et al., 2010). Since discharge peaks do not have equal base flows, the normative

discharge will be used (Fig. 2). The probability distributions of the time delays will be expressed in terms of histograms. The circumstances of the simultaneous occurrence of discharge peaks is analysed as function of the amount of discharge and the timing in the year (Fig. 3). Next, the influences of the catchment characteristics are determined on the basis of a comparison between the time shift and the catchment area, the pattern of precipitation and the stream power of the tributaries. In situations where no time shift between the peaks is observed, the water levels will be examined to determine the backwater effects of simultaneous occurrence of discharge peaks.

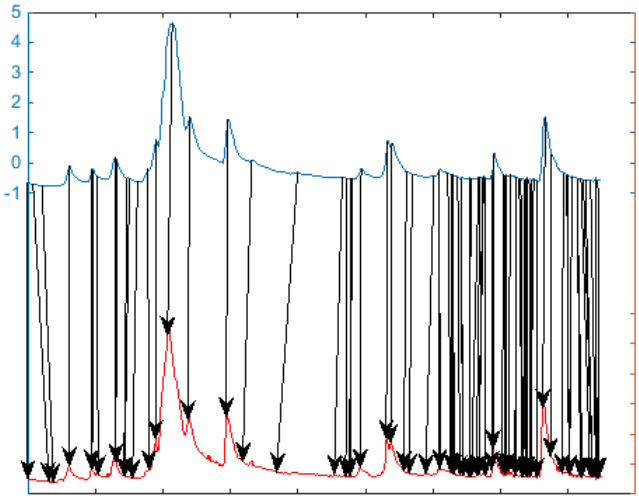

Figure 2. An example of the result of two discharge time series in which the arrows show the time shift.

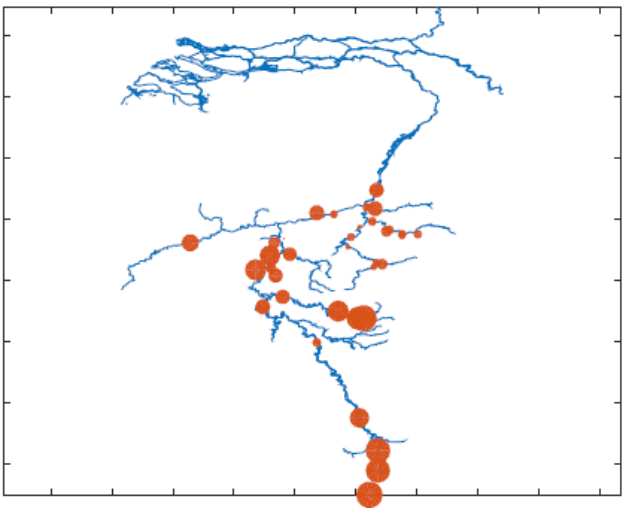

Figure 3. The travel time distribution over the Meuse basin for the 30th percent highest discharges from 1968 to 2014. The size of the dots represents the travel days to Borgharen; the smaller the size, the smaller the travel time.

Conclusions and future work

The preliminary results show that the time shifts of the tributaries of the Ardennes with the Meuse River are small. These time shifts are even smaller during high discharge peaks. The time shift will be compared with the water levels in the Meuse and its tributaries and with precipitation events in the Meuse basin.

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Floodplain rehabilitation: linking processes to patterns

V. Harezlak[a,b,*],D.C.M. Augustijn[a], G.W. Geerling[b,c], S.J.M.H. Hulscher[a] and R.S.E.W. Leuven[c]
[a] *University of Twente, Department of Water Engineering and Management, Faculty of Engineering Technology, P.O. Box 217, 7500 AE, Enschede, the Netherlands*
[b] *Deltares, Department of Freshwater Ecology and Water Quality, P.O. Box 85467, 3508 AL, Utrecht, the Netherlands*
[c] *Radboud University, Faculty of Science, Institute for Science, Innovation and Society, P.O. Box 9010, 6500 GL Nijmegen, the Netherlands*
**Corresponding author: v.harezlak@utwente.nl*

Introduction

Worldwide, floodplains are degrading at a fast rate because of human interventions: rivers have been canalized and deepened for navigation, dams have been built for hydro-power and drinking water, and embankments and channel deepening have been realized for protection of land (e.g. Buijse et al., 2002, Tockner and Standford, 2002). As such, river training has distorted the natural connection between the river and its floodplain, which has impeded the exchange of energy and material between the river and its floodplains. It is *this* limitation in river-floodplain exchanges that has resulted in the degradation of one of the world's most biologically productive and diverse ecosystems (e.g. Ward et al., 2001, Tockner and Stanford, 2002).

In the Netherlands, many floodplains of the larger (and trained) rivers are assigned as a nature protection area (LNV and V&W, 1997). Therefore, much of the former acclaimed lands for agriculture, and clay and gravel mining have been given back to nature. However, because of the disruption of energy exchange between river and floodplain, and hence the lack of the floodplain shaping processes (i.e. sedimentation and erosion), floodplain vegetation can develop to its climax stadium. This affects both water safety and nature development (Baptist et al., 2004, Geerling, 2008).

To prevent floodplains from becoming over-grown with floodplain forest (i.e. the natural climax stadium of floodplain vegetation) the landscaping force of undisturbed rivers is mimicked by implementing cyclic rejuvenation (Baptist et al., 2004, Geerling, 2008). Cyclic rejuvenation encompasses the removal of trees or grassland and even digging off parts of a floodplain. This imitates the set back of vegetation succession due to erosion in floodplains of undisturbed rivers. Because of imitating the natural set back of vegetation, cyclic rejuvenation is seen as a management strategy that is beneficial for both water safety and nature (Baptist et al., 2004).

Yet, there are still many questions on how to plan cyclic rejuvenation measures regarding the spatial development of the floodplain vegetation and its effect on especially water safety (Geerling, 2008). Moreover, in present day management, other ecosystem services floodplains have to offer, are often overlooked. This is a missed opportunity, as water safety can be combined with other ecosystem services, like biodiversity, carbon sequestration, water purification etc.

Knowledge on the dominant steering processes of the development of floodplain vegetation can be gained from the field. Plant traits, or specific plant characteristics like plant flexibility, growth form, rooting depth, timing of propagules release (e.g. Goldberg, 1990, Karrenberg et al., 2002, Navas and Violle, 2009), are sorted by environmental gradients. These environmental gradients, like inundation duration, groundwater availability, and soil texture, are the result of the processes that are taking place in a plant's environment (McGill et al., 2006). Therefore, explicitly linking plant traits in floodplains to processes via environmental gradients, will clarify how floodplain management can steer floodplain vegetation development. Furthermore, relating plant traits to ecosystem services leads to effective floodplain management, as next to water safety other ecosystem services can be made noticeable.

To aid well-conceived floodplain management, the aim of this research is therefore to develop a process based, spatially explicit model that provides insight in important steering processes of floodplain vegetation development. Additionally, this vegetation development is coupled to several ecosystem services, clarifying what floodplain management means in terms of multiple ecosystem services and not just water safety. To structure the development of the model, the following research questions are answered:

I. What processes filter floodplain vegetation traits in floodplains of undisturbed and trained rivers?
II. What modeling concepts exist that simulate the spatial and/or temporal dynamics in plant traits?

III. How can we simulate, validate and predict the development of vegetation traits in Dutch floodplains?
IV. How can we simulate, validate and predict both water conveyance and other ecosystem services in Dutch floodplains?
V. How can we upscale the floodplain model to a Dutch river stretch?

Method

To get an overview of how processes link to specific plant traits via environmental gradients, a meta-analysis is conducted. The outcome of the meta-analysis is used to build or improve a trait based model. This model will be tested on how well it selects specific plant traits under specific environmental conditions. Data for model calibration and validation comes from fieldwork. Plant traits in three Dutch floodplains will be mapped and related to floodplain processes that led to the observed plant trait composition.

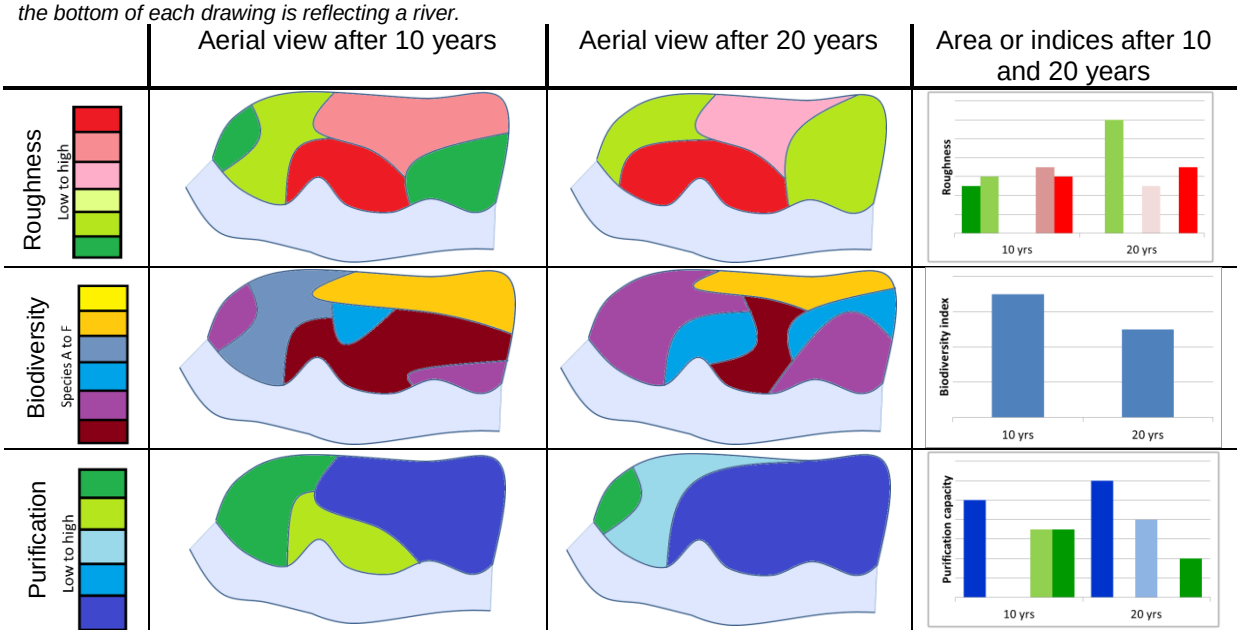
After model calibration and validation, the trait based model is elaborated with links to ecosystem services. One ecosystem service is water safety, and for that the trait based model is coupled to a flow model like Delft3D. How to relate other ecosystem services to the trait based model, is worked out in close collaboration with the RiverCare PhD candidate on Ecosystem Services.

The scale on which water safety management and nature development takes place, is (or should be) larger than the floodplain scale. It therefore makes sense to apply the model to a larger scale than just one floodplain. However, collecting detailed input data on a large scale is problematic. As such, a methodology will be developed in collaboration with the RiverCare PhD candidate on Remote Sensing to retrieve input data for the model in a labour extensive way. Additionally, the model may be adapted as well to give insight in development in vegetation patterns in a less detailed way.

Results

This PhD work leads to a tool that simulates the spatial development of vegetation traits in floodplains under autonomous conditions and management scenarios. Moreover, the model gives information on what those spatial developments imply for ecosystem services. The kind of results the model will render, is depicted in Table 1.

Table 1: Representation of the results of the trait based model. The column at the left shows the legend that is used in the figures in that row. In the second and third column model results of roughness (first row), biodiversity (second row) and water purification capacity (third row) after, respectively, 10 and 20 years from a defined year are depicted. The right column reflects the areas or index of the different classes, once again after 10 and 20 years from a defined year. Note that the light blue area at the bottom of each drawing is reflecting a river.



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Modelling bifurcations in the Tabasco coastal plain, Mexico

I.R. Lokhorst[1], C.A.M. Nooren[1], M.G. Kleinhans[1], W.Z. Hoek[1]
[1]Utrecht University, Department Physical Geography, faculty of Geosciences, P.O. Box 80.115, 3508 TC, Utrecht, The Netherlands

Introduction

The Tabasco coastal plain is situated in the Gulf of Mexico and is characterized by two of Mexico's largest rivers which flow out into the sea here. The Usumacinta and Grijalva rivers build out wave dominated deltas with very distinct beach ridges. Both the elevation data and the beach ridges show that there have been several avulsions during the Holocene. The Grijalva river however had a much higher avulsion frequency than the Usumacinta. This research aims to obtain a feasible explanation for the two most recent major avulsions we can still trace (Figure 1). It is important to obtain a decent understanding of the bifurcation dynamics in the Grijalva as the city of Villahermosa (approximately 350.000 inhabitants) is located near the nodal avulsion point and it has significant problems with the distribution of discharge and sediment over the two active branches of the Grijalva. Through using model based research combined with previously performed sedimentological research it becomes possible to get a decent understanding of the main processes at work. There is much left to discover about bifurcation dynamics. A thorough understanding of bifurcation processes can only be achieved through extensive research after these locations, but it could hopefully lead to a more sustainable and natural management of bifurcations in the future.

Methods

The bifurcation model by Kleinhans et al. (2008) was used to model the bifurcations. This model is not able to predict when a bifurcation will initiate but it can predict the evolution of the two branches once it has been initiated. The input values in the model are upstream discharge, river gradients, an arbitrary initial discharge distribution, grainsize and the initial upstream river width. These values are fairly easy obtained, and this reduces the error due to poor initial conditions.

Possible causes for the avulsions have been modelled to investigate which parameters have had the largest influence. The main causes for avulsion which were investigated were:

- Tectonics
- Sea level rise
- Confluencing streams
- Delta progradation (decreasing river gradient)

These variables have been chosen because Quaternary sedimentological research has shown that tectonics (Alexander, 1987) and sea level rise (Stouthamer and Berendsen, 2000) are of major importance for avulsions. Confluencing streams have also shown to significantly influence the outcome of a bifurcation (Kleinhans et al., 2012), and this particular area contains many confluencing streams. Delta progradation was investigated as a possible cause because the river gradients in the area are extremely low and delta progradation rates were high over the last 4000 years. In the end the outcome of the model runs was compared to the sedimentological reconstruction of the avulsions to see whether these two types of research predict the same outcome.

Results

Usumacinta

The extent of the delta has shown to be of major importance for the Usumacinta's avulsion history as the system was very sensitive to the river gradient. Because the old Usumacinta delta prograded seaward by approximately 10 km prior to the avulsion and the current branch was not yet protruding this far seaward, the currently active branch obtained a gradient advantage. The inclusion of the confluence with the Grijalva river clearly showed to enhance the avulsion process and to increase the discharge through the new branch (Figure 2). Though the presence of confluences did enhance the bifurcation process, it was not enough to cause the avulsion by itself (Figure 2).

The sediment transport of the river was sufficient to keep up with sea level rise and tectonics have shown to have little effect unless they occur locally. As detailed information on the tectonics is lacking we cannot quantify its importance.

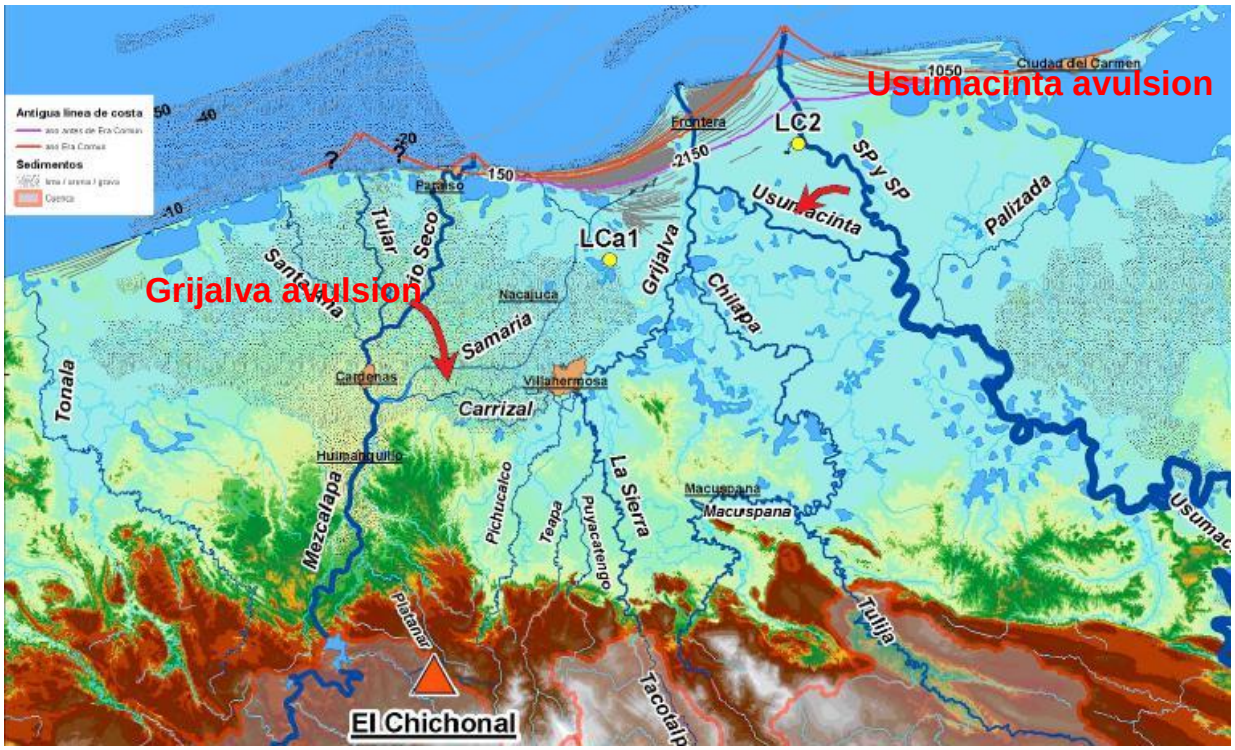

Figure 1. Map of the area including the modelled avulsions

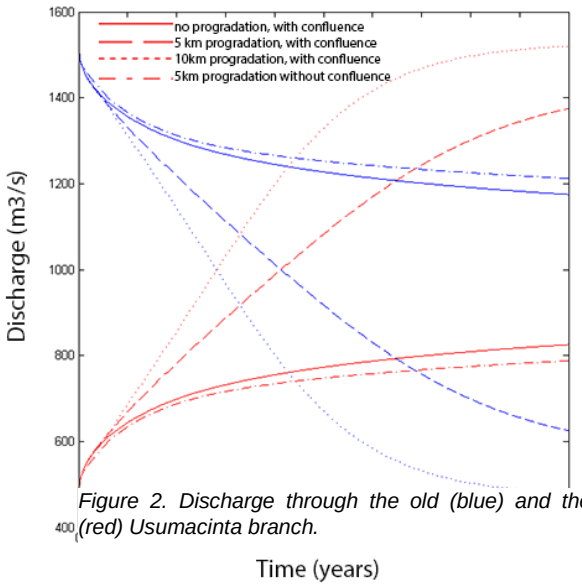

Figure 2. Discharge through the old (blue) and the new (red) Usumacinta branch.

Grijalva

The current course of the Grijalva seems to be very unfavourable in terms of gradient due to its large length (170 km), while it was 82 km for its old course. Because of this large gradient difference the only scenario which predicted avulsion was the scenario with significant tectonics in the currently active branch only. The predicted mobility of the sediment was, however, very high, which resulted in unstable model behaviour. This shows us that we might still overlook some aspects of the system that are of major importance. The higher sediment mobility and strong decrease in transport capacity might be an important trigger for the Grijalva's high avulsion frequency.

Sedimentological comparison

Detailed sedimentological and chronological research of the beach ridge system reveals that sediment volumes and progradation rates of the deltas are roughly similar to the model predictions, which shows us that the model is quite accurate.

Conclusion

The most likely reason for the Usumacinta's avulsion ~1000 year ago is that the river decreased its gradient due to delta progradation. Most likely the Grijalva river was already present at that time, and the confluence with the Usumacinta might have enhanced the avulsion process.

The only possible explanation found for the Grijalva's avulsion is tectonics. We do know that there is tectonic activity in the area, but the precise location and magnitude are not quantified. The unfavourable river gradient does explain the current problems with sediment and discharge distribution at and near the city of Villahermosa. Results also indicate that the current system is unlikely to be stable.

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Monitoring vegetation height of grassland and herbaceous vegetation with remote sensing

W.K. van Iersel*[1], M.W. Straatsma[1], E.A. Addink[1], H. Middelkoop[1]
[1] *Utrecht University, Department of Physical Geography, Faculty of Geosciences, Heidelberglaan 2, 3584 CS Utrecht*
**Corresponding author; email: W.K.vanIersel@uu.nl*

Introduction

The current change in river management strategies towards 'working with nature' allows for more spatial variation in floodplain vegetation. To derive the hydraulic roughness, ecosystem services and biodiversity from the vegetation cover an accurate vegetation classification is needed. However, this accuracy remains low for grassland and herbaceous vegetation in the current ecotope maps of the Netherlands, due to their spectral and structural similarity. Until now vegetation classification has mainly been performed with single epoch datasets using either structural or spectral characteristics of the vegetation or a combination thereof. Multi epoch data sets covering the changes over the growing season may reveal new possibilities to use vegetation type specific changes, which may be used for classification. The increased availability of Unmanned Aerial Vehicles (UAV) allows low-cost production of high resolution orthophotos and digital surface models. *The aim of this study is therefore to map temporal height profiles of grassland and herbaceous vegetation in floodplains based on multi-temporal high-spatial-resolution remote sensing data.*

Study area

The Breemwaard floodplain lies on the southern bank of the river Waal (Fig. 1). The floodplain used to have a dynamic character until the end of the 19th century due to excavation and the construction of groins (Peter and Kurstjens, 2011). Further clay and sand mining in the 20th century resulted in large pits, which developed into lakes with marshes and small riparian woods still existing today, because no recultivation occurred. Clay mining for dike reinforcement after the 1995 flood was combined with nature development in the area, which resulted in a large flood channel. The swampy northern bank of this channel was covered over time with young willow trees, which were cut down again in 2012 to reduce floodplain roughness along the major rivers (Rijkswaterstaat, 2013). Logging operations in the Breemwaard for this project will be finished in 2015.

Approximately 30% of the 116 ha area is used as hayfield and is mowed several times per year. Several parts of the floodplain are managed privately: partly grazed by ponies, managed as willow fields or reed fields. The remaining part is a nature area which is managed by the state forestry and grazed by cows and ponies. The areas fenced off from grazing activities within the nature area have developed into riparian woodland. This variability in management of the area results in a large variability of vegetation types and structures within the Breemwaard and therefore makes it a suitable area for classification experiments of riparian vegetation types.

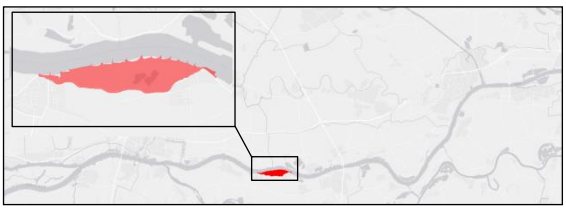

Figure 1. Location of study area along the southern bank of the river Waal.

Materials and methods

Over a period of 18 months (August 2014 – January 2015) many different types of data have been and will be collected (Fig 2). Laser scanning data have been recorded from airborne platforms (Airborne Laser Scanning; ALS) and a terrestrial mobile platform (Mobile Laser Scanning; MLS). Stereoscopic airborne imagery was collected with a UAV and field measurements of 28 plots with low (< 4 m) vegetation were carried out simultaneously with the UAV campaigns. For each plot vegetation height (H_v), vegetation density (D) and species occurrence (S) were collected.

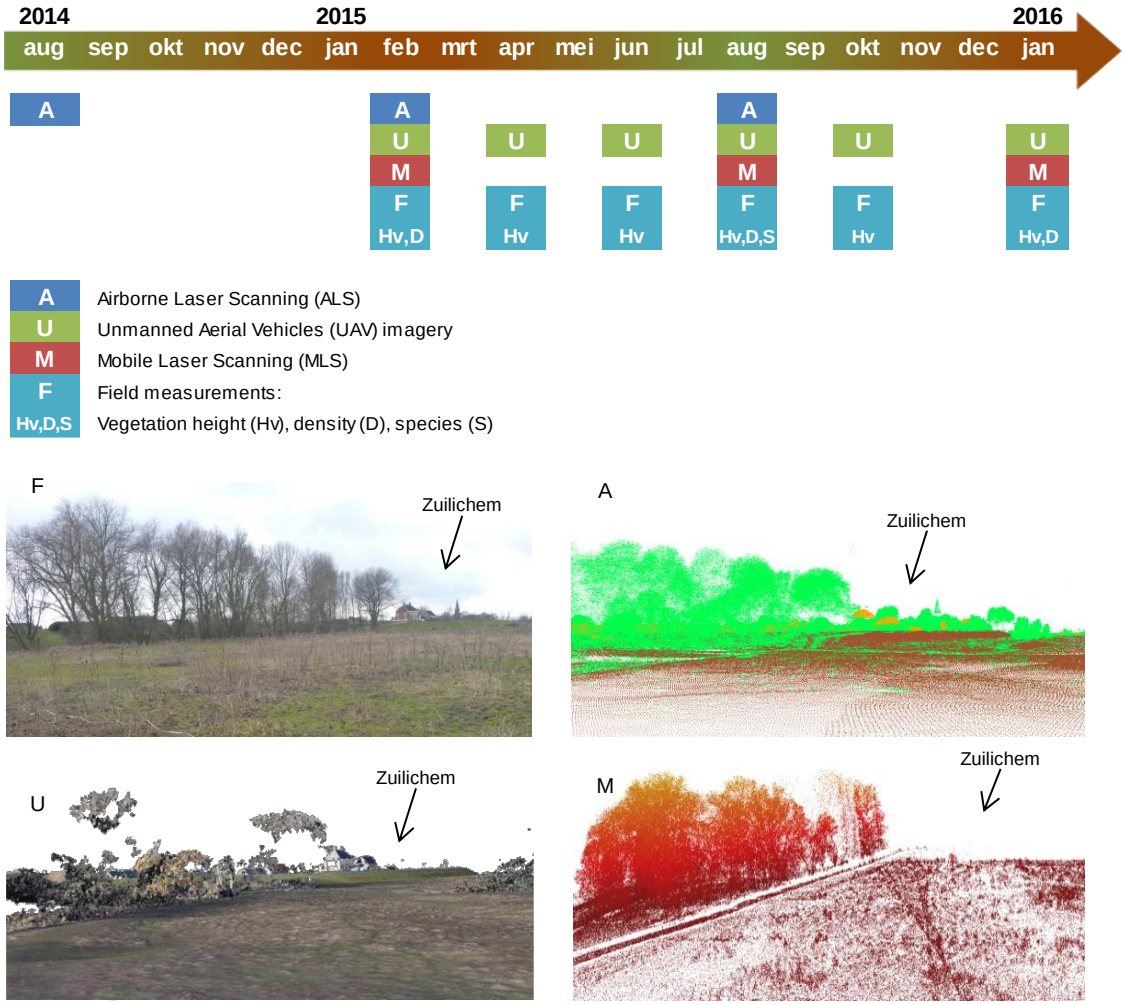

Figure 2. Timeline of data acquisition and examples of the different data types used to measure vegetation height. F) Photograph in the field towards the west. In the foreground dead teasel vegetation is visible, A) Classified ALS data of the same area (green=vegetation, brown=ground, orange=building). U) DSM derived from UAV imagery of the same area M) MLS data coloured by elevation. The village Zuilichem is not scanned with during the MLS acquisition and is therefore not visible in the background, but its location is indicated in the image.

Description of data

Airborne Laser Scanning

The acquisition of ALS data was part of two projects to collect terrain height information. The first is the national elevation model of the Netherlands (AHN), for which data acquisition takes approximately 4 winter periods. The leaf-off season is chosen to maximize the probability to measure the terrain height. The second project, which also takes a 4 year period, repeatedly collects topographic information of the riparian zone of the section of the river Waal between Lent and Giessen. The ALS data were delivered as a georeferenced point cloud with GPS-time at first entry, xyz coordinates, intensity values, scan angle and return number. The AHN data was also available in full-waveform format, but this was not used in this study, to keep data types consistent. Specifications of the ALS data can be found in Table 1.

Table 1. Specifications of ALS and MLS laser scanners (RIEGL, 2011; Riegl, 2015)

	ALS data	MLS data
Data provider	Fugro Geospatial BV	Utrecht University
Scanner	Riegl LMS-780	Riegl VZ-400
Laser wavelength	near infrared	near infrared
Point spacing	0.25-0.29 m	highly variable
PRR	up to 400 kHz	up to 300 kHz
Number of targets per pulse	unlimited (full waveform)	Up to 4
File format	LAS 1.2	LAS 1.2
Acquisition height	~460 m	~2.20 m

*PRR: laser pulse repetition rate

Mobile Laser Scanning

The laser data from the mobile platform were collected with a laser scanner mounted on top of a four-wheel drive car. The laser scanner was positioned either looking to the left or right side of the car while driving, scanning the surface and vegetation obliquely. The MLS data were delivered as a georeferenced point cloud

with GPS-time at first entry, xyz coordinates, intensity values, and return number. MLS point density varies strongly, with the highest values close to the vehicle due to divergence of the laser array. Up to four returns were recorded from a single MLS pulse. Further specifications of the MLS data are presented in Table 1.

UAV imagery

Airborne imagery was collected using the Swinglet CAM as UAV. Specifications of the UAV imagery can be found in Table 2. The UAV is mounted with a GPS receiver, altimeter, wind meter and a camera. The camera is electronically triggered by the autopilot system to acquire images at the required positions. All images were georeferenced by the on-board GPS of the UAV at the moment they are taken by the camera. Because RAW-format imagery was not possible with this camera, no radiometric information is available of the images after compression into JPEG-format. The true colour imagery was used to produce the point-based DSMs.

Table 2. Specifications of the UAV imagery

	UAV imagery
Data provider	HiView
UAV	Swinglet CAM by Sensefly
Camera	Canon IXUS 125 HS
Sensor resolution	16 megapixel
Ground resolution	5 cm
File format	JPEG
Acquisition height	~150 m

Ground control of UAV imagery

Ground control points for georeferencing of the DSMs and orthophotos were obtained by 40 white vinyl markers. There were distributed over the study area and were geolocated with a dGPS, resulting in a horizontal accuracy of 0.015 m, and a vertical accuracy 0.02 m.

Field data

Field data were collected over 28 plots with low vegetation: grassland, herbaceous vegetation and helophytes. Per plot average vegetation height ($H_{v,F}$) was collected. Additionally, vegetation density was measured in February and August (Fig.1) and the vegetation species were investigated by means of a vegetation survey in August. $H_{v,F}$ was determined from 30 random maximum height measurements of plants or stems of senescent plants within the plot. Vegetation density was determined by multiplying average stem diameter with the number of stems per square meter. Stem diameter was measured for the same stems as used for $H_{v,F}$. Stem density was estimated by a counting of the number of stems within a 2 by 2 m representative subplot. Vegetation surveys were performed according to the Tansley method (Tansley, 1946). This scale uses nine classes to indicate the frequency of occurrence of the species in the plot. The vegetation surveys were used to derive vegetation types of the plots on a more aggregated level than specific species, but on a more detailed level than often used 'grassland', 'herbaceous vegetation' or 'helophytes' classes.

Data processing

ALS and MLS data

In order to predict H_v from the ALS and MLS data the height difference between the ground points and vegetation points has to be calculated ($H_{v,ALS}$ and $H_{v,MLS}$ respectively). First, the point data were filtered to eliminate outlying points. Second, the points were classified into either vegetation or ground, because no other objects occurred in the field plots of interest. The ground points were used to normalize the vegetation points into a point cloud with height above the ground. Statistics of the normalized vegetation points were regressed against field reference data to predict $H_{v,F}$. For processing of the laser data the software LAStools was used (Isenburg, 2014).

UAV imagery

The stereoscopic photos were used to obtain terrain height information. The photos were processed into point-based Digital Surface Model (DSM) of the floodplain using a Structure from Motion (SfM) workflow (Lucieer et al., 2014). The SfM processing was performed in the commercial software package Agisoft PhotoScan Professional version 1.1 (Agisoft, 2014). The specific algorithms implemented in Photoscan are not detailed in the manual, however, a description of the SfM procedure in Photoscan and commonly used parameters are described in Verhoeven (2011).

When the surface is covered with vegetation of a negligible height (winter conditions) the derived surface is assumed to represent the ground level and the derived model is a digital elevation model (DEM). In spring and summer conditions, however, the surface represents the top of the vegetation and the derived model is a digital surface model (DSM). $H_{v,UAV}$ can be derived by calculating the difference between the DEM and DSM. We normalized the DSMs (nDSM) of April, June and August using the DEM from February. Statistics of the normalized nDSM were regressed against field reference data to predict $H_{v,F}$.

Validation of RS data with field data

The data analyses of the ALS, MLS, and UAV data resulted in normalized points clouds, with each point classified as either representing the vegetation or the ground surface. To predict H_v from these data, the following statistics were computed for the vegetation points within each field plot. These describe the vertical point distribution of vegetation points above the terrain: mean, standard deviation, and percentiles (50, 70, 90, 95%). Regression analyses were performed with the $H_{v,F}$ as the dependent variable and the statistics as independent variables. From these analyses regression equations have been obtained describing the relation between $H_{v,F}$ and the vegetation height related statistics computed from the ALS, MLS, and stereoscopic photos.

The method above was repeated for each ALS, MLS, and UAV campaign, resulting in a series of predicted vegetation heights over the growing season. This enabled to construction of temporal vegetation height profiles.

Results

Preliminary results of the ongoing data collection will be shown as well as the first results of regression analysis indicating the possibilities for prediction of height profiles derived from RS data.

Conclusion

The aim of this study is to map temporal height profiles of low vegetation in floodplains based on multi-temporal high-spatial-resolution remote sensing data. Over a period of 18 months different types of data have been and will be collected. The current data collection consists of Airborne and Mobile Laser Scanning data, UAV imagery and field measurements of vegetation. The relation between vegetation height (H_v) derived from the remote sensing data and H_v measured in the field is explored.

If a significant relation can be established, UAV or LiDAR remotely sensed H_v may be used during the classification of low vegetation at floodplain scale (< 1 km scale). In addition, the monitoring of H_v over time in this study may be indicative for the best time to collect the remote sensing data for mapping vegetation height in floodplains.

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Wageningen University
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