Flood Risk Science and Management

Edited by

Gareth Pender
Hazel Faulkner

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Preface

Approaches to avoid loss of life and limit disruption and damage from flooding have changed significantly in recent years. There has been a move from a strategy of flood defence to one of flood risk management. Flood risk management includes flood prevention using hard defences, where appropriate, but also requires that society learns to live with floods and that stakeholders living in flood prone areas develop coping strategies to increase their resilience to flood impacts when these occur. This change in approach represents a paradigm shift which stems from the realisation that continuing to strengthen and extend conventional flood defences is unsustainable economically, environmentally, and in terms of social equity. Flood risk management recognises that a sustainable approach must rest on integrated measures that reduce not only the probability of flooding, but also the consequences. This is essential as increases in the probability of inundation are inevitable in many areas of the world due to climate change, while socio-economic development will lead to spiralling increases in the consequences of flooding unless land use in floodplains is carefully planned.

Recognizing the need for research to support this shift, funders of flood risk management research in the UK created the Flood Risk Management Research Consortium (FRMRC), a multi-institutional, multi-disciplinary consortium tasked with increasing the understanding of flooding by generating new and original science, to support improved flood risk management. The portfolio of activities included:

- the short-term delivery of tools and techniques to support accurate flood forecasting and warning, improvements to flood management infrastructure and reduction of flood risk to people, property and the environment;
- the establishment of a programme of high quality science to enhance understanding of flooding and improve society's ability to reduce flood risk through the development of sustainable flood management strategies.

The core content for this volume has been provided by members of the FRMRC. In addition, we have broadened the range of expertise by drawing on the international research community in flood management. Our intention is to provide an extensive and comprehensive synthesis of current international research in flood management, thereby, providing a multi-disciplinary reference text covering a wide range of flood management topics.

The book authors are at the very highest position in academic institutions researching Flood Risk Science and Management in the UK and elsewhere. The contents are organised into seven parts. Part 1 of the text develops a scene-setting overview of contemporary scientific and socio-economic challenges, drawing largely on the situation in Europe and the UK in particular. In Part 2, land-use management is explored as a strategic approach to flood risk reduction. Flood frequency changes consequent upon land-use modifications under current climatic and socio-economic 'futures' are explored, the multi-scale impacts of land management on flooding are developed further in the case study context of FRMRC's Pont Bren study area, a subcatchment of the river Severn in mid-Wales, UK (Chapters 2 & 3). In Chapter 4, the coastal management strategies of managed retreat, managed realignment and restoration are reviewed as approaches to coastal flood risk. The issues associated with sediment management in flood models and in management schemes are explored in Chapter 5, and flood defence and asset appraisal reviewed in Chapter 6.

In Part 3, flood forecasting and the issuing of warnings are both considered from a technical perspective. Chapters 7 and 8 look at advances in remote sensing; in Chapter 7 in relation to precipitation estimation using radar, and in Chapter 8 in relation to real-time flood forecasting. The challenges of updating forecasts in real-time is explored in Chapter 9, and the problems associated with coupling rainfall and run-off models are considered in Chapter 10.

Flood modelling, and the modelling of flood mitigation effects is the focus of Part 4. Chapter 11 covers data utilisation for modelling purposes, and Chapters 12 and 13 develop the algorithm for 1D-2D modelling in a range of settings. The tools available for handling uncertainties in models are outlined in Part 5. The risk-based approach is further developed in the context of asset management in Chapter 15, and in coastal modelling in Chapter 16.

In Part 6, policy and planning are both addressed from a predominantly socio-economic perspective. Governance issues (Chapter 17), the involvement of stakeholders in practice and management (Chapter 18), and the design of effective ways to target flood risk communications (Chapter 19) are considered first. Some of the psycho-social dimensions of Flood Risk Management are explored in Chapter 20, and health impacts of flooding in Chapter 21. The remaining chapters in Part 7 trace key case studies from a range of international settings.

This text covering *Flood Risk Science and Management* therefore provides an extensive and comprehensive synthesis of current research in flood management; developing a multi-disciplinary reference text covering a wide range of flood management topics. Its targeted readership is the international research community (from research students through to senior staff), as well as flood management professionals, such as engineers, planners, government officials and those with flood management responsibility in the public sector. By using the concept of case study chapters, international coverage is given to the topic, ensuring a world-wide relevance.

ACKNOWLEDGEMENTS

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Gareth Pender and Hazel Faulkner
September 2010

Contributors

TONY ANDRYSZEWSKI *Policy Manager, Environment Agency, Aztec House, Bristol, UK*

RODOLFO ARADAS, PhD *Halcrow Group Ltd, Buenos Aires, Argentina*

BARNEY N. AUSTIN, PhD *PE Director of Hydrologic Services, INTERA, Inc., Austin, TX, USA*

GEOFFREY L. AUSTIN, MSC, PhD, FRSNZ *Professor of Geophysics, Physics Department, University of Auckland, New Zealand*

CAROLINE BALLARD, ME *Research Assistant, Department of Civil and Environmental Engineering, Imperial College London, UK*

PAUL D. BATES, PhD *Professor of Hydrology, School of Geographical Sciences, University of Bristol, UK*

GEOFF BAXTER PhD *Research Scientist - Flooding and Communities, Research, Monitoring and Innovation Evidence Directorate, Environment Agency, Cardiff, Wales, UK*

KEITH BEVEN, PhD *Professor of Hydrology, Lancaster Environment Centre, Lancaster University, Lancaster, LA1 4YQ, UK*

DAVID S. BIEDENHARN, PhD *Principal Investigator, Biedenharn Group, LLC, Vicksburg, MS 39180, USA*

JOHN BLANKSBY, DMS, CENG MICE *Research Fellow, Department of Civil and Structural Engineering, University of Sheffield, UK*

ANDREW BROOKES, PhD, FCIWEM, MIEMA, C.WEM, CENV, CSCI, CGEOG *Divisional Director, Jacobs Engineering UK Ltd, 1180 Eskdale Road, Winnersh, Wokingham, RG14 5TU, UK*

NATALIYA S BULYGINA, PhD *Research Associate, Department of Civil and Environmental Engineering, Imperial College London, UK*

IAN D. CLUCKIE, MSC., PhD, FRENG, CENG, FICE, FCIWEM, FRSA, FRMETSOC *Professor and Pro-Vice-Chancellor (Science and Engineering), Swansea University, Wales, UK*

TOM J. COULTHARD, PhD *Professor of Physical Geography, University of Hull, Hull, HU6 7RX, UK*

MARK DIXON, FIMS, MBE *Independent, West Mersea, Essex*

SLOBODAN DJORDJEVIĆ, PhD *Associate Professor of Hydraulic Engineering, Centre for Water Systems, College of Engineering, Mathematics and Physical Sciences, University of Exeter, UK*

JOHN EWEN, PhD *School of Civil Engineering and Geosciences, Newcastle University, Newcastle upon Tyne, UK*

HAZEL FAULKNER, MSC, PhD *Professor of Environmental Management, Flood Hazard Research Centre, Middlesex University, UK*

LORNA FEWTRELL, PhD, FRSPH *Centre for Research into Environment and Health, Aberystwyth University, Wales, UK*

ZOE FROGBROOK, PhD *Advisor: Soils and Forestry, Environment Agency Wales, UK. Formerly at Centre for Ecology and Hydrology, Environment Centre Wales, Bangor, Wales, UK*

GERALD E. GALLOWAY, PhD, PE *Professor of Engineering, Glenn L. Martin Institute, University of Maryland, USA*

STANFORD GIBSON, PhD *Research Hydraulic Engineer, Hydrologic Engineering Center, Davis, CA 95616, USA*

NIGEL J.D. GRAHAM, MSC, PhD, DIC, SCD, FICE, FICHEME, FCIWEM, CENG, CSCI, CENV *Professor of Environmental Engineering, Department of Civil and Environmental Engineering, Imperial College, London, UK*

COLIN GREEN, MA *Professor of Geography, Flood Hazard Research Centre, Middlesex University, UK*

JIM W. HALL, PhD, CENG, FICE *Professor of Earth Systems Engineering, School of Civil Engineering and Geosciences, Newcastle University, Newcastle upon Tyne, UK*

J. BARRIE HOLDEN, CCHEM, MRSC *Innovation Programme Manager, Anglian Water Services Ltd, Peterborough, UK*

JOSÉ HORRILLO-CARABALLO, PhD *Research Fellow, School of Engineering, University of Plymouth, UK*

BETHANNA M JACKSON, PhD *Lecturer in Hydrology and Water Resources, School of Geography, Environment and Earth Sciences, Victo, ria University of Wellington, New Zealand. Formerly at Imperial College London, UK*

DAVID KAY, PhD *Professor of Environment and Health, Centre for Catchment and Coastal Research and Centre for Research into Environment and Health, Hydrology and River Basin Dynamics Research Group, IGES, Aberystwyth University, Wales, UK*

MALIK FIDA ABDULLAH KHAN *CEGIS, Dhaka, Bangladesh*

JONATHAN LAWRY, PhD *Professor of Artificial Intelligence, Department of Engineering Mathematics, University of Bristol, Queen's Building, University Walk, Bristol, UK*

CHARLES D. LITTLE JR, PE *Research Hydraulic Engineer, US Army Corps of Engineers, Engineer Research & Development Center, Vicksburg, MS 39180, USA*

GAVIN LONG, PhD *Research Associate, Department of Civil Engineering, University of Nottingham, UK*

LUKE LOVELL, MSC *Halcrow, Swindon, UK*

ČEDO MAKSIMOVIĆ, PhD *Professorial Research Fellow, Centre for Urban Water, Department of Civil and Environmental Engineering, Imperial College London, UK*

MILES R. MARSHALL, PhD *Teaching Fellow in Physical Geography, School of Geography, Environment and Earth Sciences, Victoria University of Wellington, New Zealand. Formerly at Imperial College London, UK*

DAVID C. MASON, PhD *Reader, NERC Environmental Systems Science Centre, University of Reading, UK*

MICHAEL J. MAWDESLEY, PhD *Associate Professor, Department of Civil Engineering, The University of Nottingham, UK*

SIMON McCARTHY, PhD *Researcher, Flood Hazard Research Centre, Middlesex University, UK*

DANIEL R. McCULLOCH *formerly Department of Engineering Mathematics, University of Bristol, Queen's Building, University Walk, Bristol, UK*

NEIL MCINTYRE, PhD *Reader, Department of Civil and Environmental Engineering, Imperial College London, UK*

DAVID M. MOONEY, PhD, PE *Program Engineer, San Joaquin River Restoration Project, Bureau of Reclamation, Sacramento, CA 95825, USA*

AINUN NISHAT, PhD *IUCN-World Conservation Union, Dhaka, Bangladesh*

BUSHRA NISHAT *Institute of Water Modelling, Bangladesh*

SYLVAIN NÉELZ, PhD *Former Lecturer, School of the Built Environment, Heriot-Watt University, Edinburgh, Scotland, UK*

ENDA O'CONNELL, PhD *Professor of Water Resources Engineering, School of Civil Engineering and Geosciences, Newcastle University, Newcastle upon Tyne, UK.*

GREG O'DONNELL, PhD *Catchment Change Network, School of Civil Engineering and Geosciences, Newcastle University, Newcastle upon Tyne, UK*

ADRIÁN PEDROZO-ACUÑA, PhD *Investigador, Instituto de Ingeniería, Universidad Nacional Autónoma de México, Mexico*

GARETH PENDER, PhD, CENG, FICE, FCIWEM, FRSE *Professor of Environmental Engineering, School of the Built Environment, Heriot-Watt University, Edinburgh, Scotland, UK*

EDMUND PENNING-ROWSELL, MA, PhD *Professor of Geography, Flood Hazard Research Centre, Middlesex University, UK*

NICHOLAS J. RANDON *Artificial Intelligence Group, Department of Engineering Mathematics, University of Bristol, BS8 1TR, UK*

DOMINIC E. REEVE, PhD, CMATH, FIMA, FICE *Professor of Coastal Dynamics, School of Engineering, University of Plymouth, UK*

BRIAN REYNOLDS, PhD *Senior Water Quality Scientist, Centre for Ecology and Hydrology, Environment Centre Wales, Bangor, Wales, UK*

ADRIAN J. SAUL, PhD, FICE *Yorkshire Water Chair of Water Engineering, Pennine Water Group, University of Sheffield, Department of Civil and Structural Engineering, Sheffield, UK*

PAUL SAYERS, PhD *Principal Partner, Sayers and Partners, Watlington, UK*

GUY J-P. SCHUMANN, PhD *GWR Fellow, School of Geographical Sciences, University of Bristol, UK*

COLIN SCOTT, MSC *ABP Marine Environmental Research (ABPmer), UK*

PAUL SHUCKSMITH, MSC *Graduate student, Atmospheric Physics Group, University of Auckland, New Zealand*

JONATHAN SIMM *Hydraulics Research Wallingford Ltd, Wallingford, UK*

KEREN SMITH, PhD *Centre for Research into Environment and Health, Aberystwyth University, Wales, UK*

PHILIP J. SOAR, PhD, CENV, CSCI, CWEM, MCIWEM *Senior Lecturer in Physical Geography, University of Portsmouth, Portsmouth, PO1 3HE, UK*

LUKE SUTHERLAND-STACEY, MSC *Graduate student, Atmospheric Physics Group, University of Auckland, New Zealand*

SUE TAPSELL, MA *Head of Flood Hazard Research Centre, Middlesex University, London, UK*

OWEN TARRANT *Environment Agency, Bristol, UK*

COLIN R. THORNE, PhD *Professor and Chair of Physical Geography, University of Nottingham, Nottingham, NG7 2RD, UK*

IAN TOWNEND, CENG, CMARSCI, FICE, FIMAREST, MCIWEM *HR Wallingford, Howbery Park, Wallingford; and University of Southampton, UK*

SYLVIA TUNSTALL, MA *Senior Researcher, Flood Hazard Research Centre, Middlesex University, UK*

MICHAEL WALLIS *Hydraulics Research Wallingford Ltd, Wallingford, UK*

MARCO J. VAN DE WIEL, PhD *University of Western Ontario, London, Ontario, N6A 5C2, Canada*

NICHOLAS P. WALLERSTEIN, PhD *Post Doctoral Research Associate, School of the Built Environment, Heriot-Watt University, Edinburgh, EH14 4AS, UK*

CHESTER C. WATSON, PhD *Principal Investigator, Biedenharn Group, LLC, 1917 Cottonwood Point Drive, Fort Collins, CO 80525, USA*

HOWARD S. WHEATER, MA, PhD, FRENG, FICE, CENG *Professor of Hydrology, Department of Civil and Environmental Engineering, Imperial College, London, UK*

JON WICKS, PhD *Halcrow, Swindon, UK*

DUNCAN WISHART, MSC, PhD, MCIWEM, CWEM, CENV, CSCI *Principal Geomorphologist, Jacobs Engineering UK Ltd, 1 City Walk, Leeds, LS11 9DX, UK*

NIGEL G. WRIGHT, BSC, PhD, CENG, FHEA, FASCE, FICE *Professor of Water and Environmental Engineering, University of Leeds, UK*

PETER C. YOUNG, MSC (LOUGHBOROUGH), MA, PhD (CANTAB), WHF, CENG *Professor Emeritus, Lancaster Environment Centre, University of Lancaster, Lancaster, LA1 4YQ, UK; Adjunct Professor, Fenner School of Environment and Society, Australian National University, Canberra, Australia; Visiting Professor, School of Electrical Engineering & Telecommunications, University of New South Wales, Sydney, Australia*

Foreword

Flooding is an important issue in the UK; over £200 billion worth of assets are at risk around British rivers and coasts, and those risks are likely to increase in the future due to climate change. To assist in managing these risks the joint Defra/EA Flood and Coastal Erosion Risk Management Research and Development Programme (FCERM) aims to ensure the development of high-quality R&D outputs that provide the evidence required for sustainable flood and coastal erosion risk management policy, process and delivery.

In 2004, the programme managers entered into an agreement with the Engineering and Physical Sciences Research Council (EPSRC), the Natural Environment Research Council (NERC), the Scottish Parliament and UK Water Industry Research (UKWIR), to fund the interdisciplinary Flood Risk Management and Research Consortium. The rationale behind this innovative joint funding arrangement was to combine the strengths of fundamental and near-market researchers and research philosophies in a truly multi-disciplinary programme. The research portfolio was designed to address medium-term issues in flood science and engineering, while being consistent with the objectives of the overall FCERM programme.

This volume is underpinned by the outcomes from the consortium's research programme and I am delighted to provide this foreword. The editors have been successful in collecting together key research papers from consortium members and their international collaborators, to produce a monograph of important scientific findings set within a multi-disciplinary context.

Flood Risk Science and Management therefore supports the goal of improved flood and coastal erosion risk management in both a UK and an international setting. This book makes a significant contribution to the Environment Agency's task of improving definitions of flood risk and meeting the challenges of defining and coping with the uncertainties that flooding brings for UK flood managers.

David Rooke MBE, BSc(Hons), CEng, FICE, FCIWEM

Acting Director of Flood and Coastal Risk Management
Environment Agency
Rio House
Waterside Drive
Aztec West
BRISTOL
BS32 4UD

Acronyms/Glossary of terms

AAD	Annual Average Damage
ADCP	Acoustic Doppler Current Profiler
ADI	Alternating Direction Implicit
AMit	Asset Management IT System
AOD	Above Ordnance Datum
AR model	Auto Regressive model
ARMA modelling	Auto Regressive Moving Average modelling
ASAR	Advanced Synthetic Aperture Radar
ASMITA	Aggregated Scale Morphological Interaction between a Tidal Basin and the Adjacent Coast
AUDACIOUS	Adaptable Urban Drainage – addressing Change in Intensity, Occurrence and Uncertainty in Stormwater
BaRE	Bayesian Recursive Estimator
BGS	British Geological Survey
BODC	British Oceanographic Data Centre
CAESAR	Cellular Automaton Evolutionary Slope and River Model
CCA	Canonical Correlation Analysis
CD	Chart Datum
CEH	Centre for Ecology and Hydrology
CES	Conveyance Estimation System
CFD	Computational Fluid Dynamics
CFX	Commercial computational fluid dynamics programme used to simulate fluid flow in a variety of applications.
CI	Condition Index
CIRIA	Construction Industry Research & Information Association
CIWEM	The Chartered Institution of Water & Environment Management
CSO	Chief Scientist Office
DBM	Data Based Mechanistic
DEFRA	Department of Environment, Food & Rural Affairs
DEM	Digital Elevation Model
DEM	Dynamic Emulation Model
DETR	Department of Environment, Transport & the Regions
DOS	Disk Operating System
DPSIR	Drivers, Pressures, States, Impacts, Responses
DSD	Drop Size Distribution
DSM	Digital Surface Model
DTM	Digital Terrain Model
DYNIA	DYNamic Indentifiability Analysis
EA	Environment Agency

EAD	Expected Annual Damage
ECMWF	European Centre for Medium range Weather Forecasting
EKF	Extended Kalman Filter
EnKF	Ensemble Kalman Filter
ENO	Essentially Non-Oscillatory
ENVISAT (satellite)	Environmental satellite
EO	Earth Observation
EOF	Empirical Orthogonal Function
ERS-2	European Remote Sensing Satellite
ESA	European Space Agency
FEPA	Food & Environmental Protection Act
FI	Failure Likelihood Index
FRF	Field Research Facility
FRMRC	Flood Risk Management Research Consortium
FRSM	Rapid Flood Spreading Method
FTT	French Tide Table
GAs	Genetic Algorithms
GCM	Generator-Coordinate-Method
GIS	Geographic Information System
GLUE	Generalised Likelihood Uncertainty Estimation
GNU	Computer operating system
GPS	Global Positioning System
GRW	Generalised Random Walk
GSM	Global System for Mobile (communications)
HEC-RAS	Hydraulic Engineering Centre – River Analysis System
HMA	Heterogeneous Missions Accessibility
HRU	Hydrological Response Unit
HYMOD model	A 5-parameter conceptual rainfall runoff model.
Hypsometry	The establishment of elevations or altitudes
ICESat (satellite)	Ice, Cloud and Land Elevation satellite
InHM	Integrated Hydrological Model
INS	Inertial Navigation System
InSAR	Interferometric Synthetic Aperture Radar
IR	Infrared
IV	Instrumental Variable
JFLOW	A multiscale two-dimensional (2D) dynamic flood model
KF	Kalman Filter
LAT	Lowest Astronomical Tide
LDT	Linguistic Decision Tree
LID	Low impact development
LiDAR	Light Detection & Ranging Data

LISFLOOD – FP	A 2-dimensional hydrodynamic model specifically designed to simulate floodplains inundation in a computationally efficient manner over complex topography
LISFLOOD (model)	a GIS-based distributed model for river basin scale water balance and flood simulation
LSEs	Limit State Equations
LSPs	Land Surface Parameterisations
LWC	Liquid Water Content
MCMC	Markov Chain Monte Carlo
MCS	Monte Carlo Simulation
MDSF2	Modelling Decision Support Framework
MIKE SHE	Dynamic, user-friendly modelling tool that can simulate the entire land phase of the hydrologic cycle
MISR	Multiangle Imaging Spectroradiometer
ML	Maximum likelihood
MLE	Multiple Linking Elements
MODIS	Moderate Resolution Imaging Spectroradiometer
MOPS	Moisture Observation Pre-processing System
MUSCL	Monotonic Up-Stream Centred Schemes for Conservation Laws
NaFRA	National Flood Risk Analysis
NCAR	National Centre for Atmospheric Research
NERC	Natural Environment Research Council
NFCDD	National Flood & Coastal Defence Database
NFFS	National Flood Forecasting System
NOAA	National Oceanic & Atmospheric Administration
NWP	Numerical Weather Prediction
OMS	Object Modelling System
Open FTA	Open Fault Tree Analysis
PAMS	Performance-based Asset Management System
PF	Particle Filter
PF	Performance Features
PV (damage)	Present Value
QPBRRM (model)	Quasi Physically-Based Rainfall Runoff model
QPF	Quantitative Precipitation Forecasting
RAFT	Risk Assessment Field-based Tool
RASP	Risk Assessment of Flood & Coastal Defence for Strategic Planning
RCM	Relative Confidence Measure
REAS	River Energy Audit Scheme
REW	Representative Elementary Watershed
RHI (scan)	Range Height Indicator
RHS	Royal Horticultural Society

RIV	Refined Instrumental Variable
RLS	Recursive Least Squares
RMS	Root Mean Square
RMSE	Root mean squared Error
RPE	Recursive Prediction Error
RW	Random Walk
SAC	Special Area of Conservation
SAR	Synthetic Aperture Radar
SDPR	State Dependent Parameter Regression
SEPA	Scottish Environment Protection Agency
SHE	Système Hydrologique Européen
SHETRAN	A three-dimensional, coupled surface/subsurface, physically-based, spatially-distributed, finite-difference model for coupled water flow, multifraction sediment transport and multiple, reactive solute transport in river basins
SIAM	Sediment Impact Assessment Model
SIPSON (model)	Simulation of Interaction between Pipe flow and Surface Overland flow in Networks
SLURP (model)	Semi-distributed Land Use-based Runoff Processes
SMHI	Sveriges Meteorolgiska och Hydrologiska Institut (Swedish)
SOBEK	1-dimensional and 2-dimensional instrument for flood forecasting, drainage systems, irrigation systems, sewer overflow etc
SRTM	Shuttle Radar Topography Mission
SSSI	Site of Special Scientific Interest
SUDS	Sustainable Drainage Systems
SWMP	Surface Water Management Plans
SWOT	Surface Water Ocean Topography
TF	Transfer Function
TIN	Triangular Irregular Network
TUFLOW software	a one-dimensional and two-dimensional flood and tide simulation software
TVD	Total Variation Diminishing
TVP	Time Variable Parameter
UIM	Urban Integrated Model
UKF	Unscented Kalman Filter
UKWIR	UK Water Industry Research
USACE	United States Army Corps of Engineers
VPR	Vertical Reflectivity of Precipitation
VPR	Vertically Pointing Radar
VRP	Vertical Reflectivity Profile
WaPUG	The Urban Drainage Group of the CIWEM
WRIP	Weather Radar Information Processor
WSP	Whole Systems Partnership

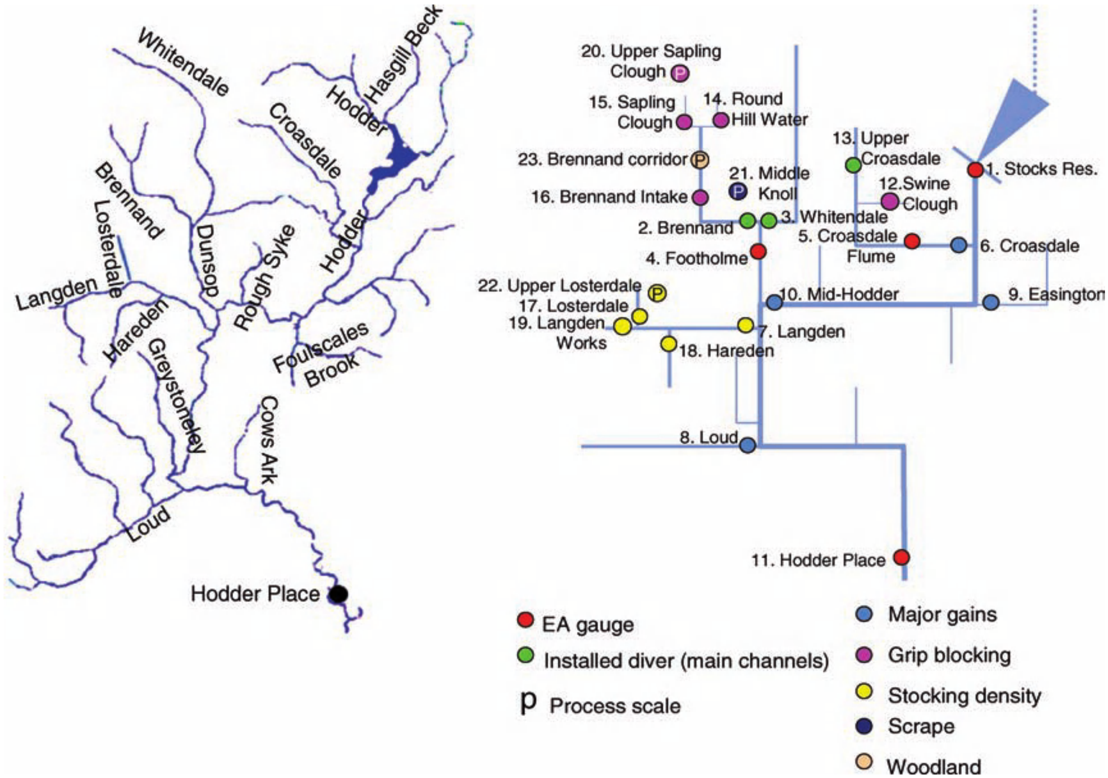

Fig. 2.6 Multiscale nested experiment in the Hodder. EA, Environment Agency.

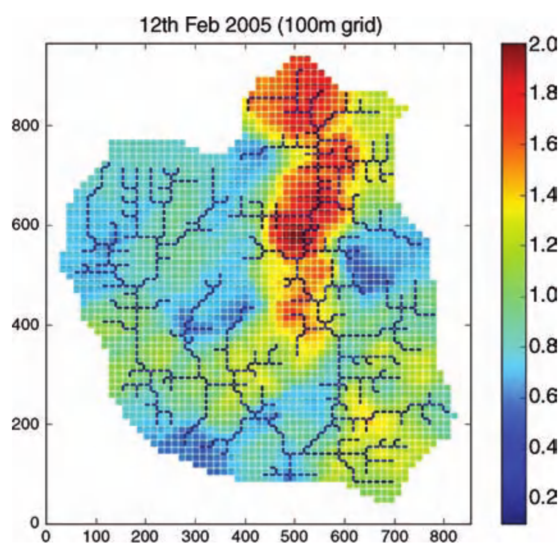

Fig. 2.8 Vulnerability map for Dunsop catchment (26 km^2) created using adjoint modelling, showing sensitivity of flood peak flow to change in Strickler coefficient.

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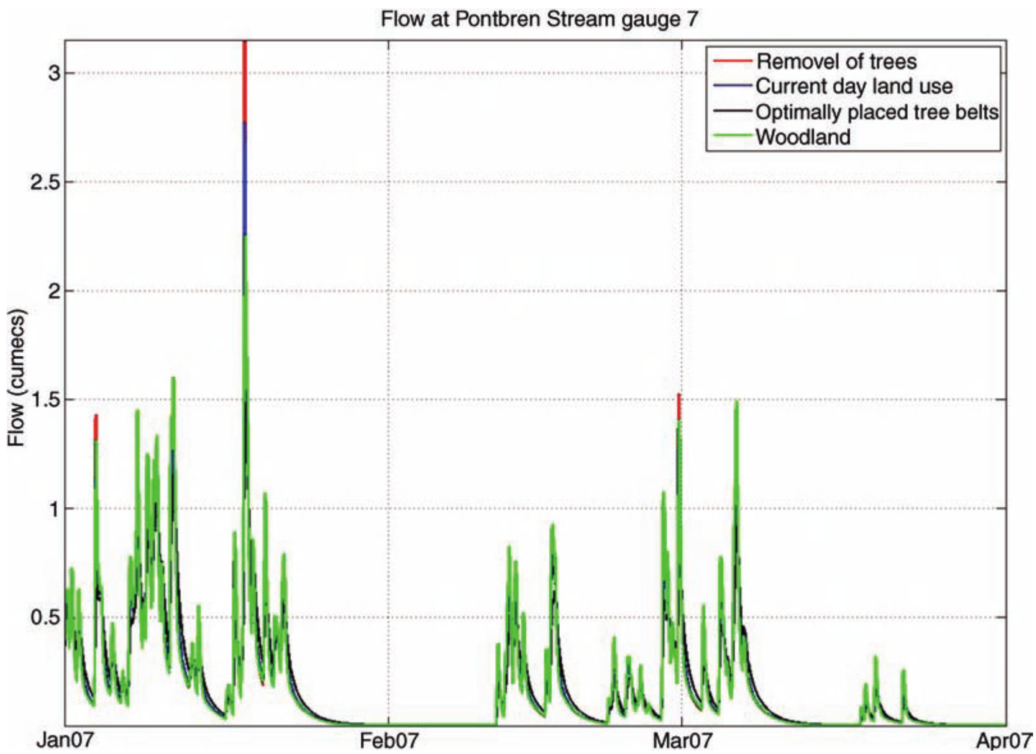


Fig. 3.11 Catchment-scale response for a 4-km^2 Pontbren subcatchment for current land use and a set of scenarios: 1990s intensification, further addition of tree shelter belts, and full aforestation.

Fig. 4.3 Initial flooding of the first scheme on the Wallasea site, where the design made use of the existing field drains.

Fig. 4.4 Example of dendritic network design for a more extensive realignment on the Wallasea site.

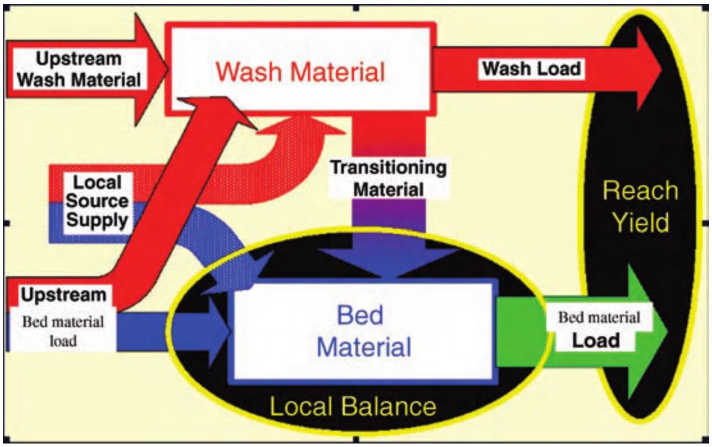


Fig. 5.8 Flow diagram illustrating how SIAM (Sediment Impact Analysis Method) accounts for bed material and wash load dynamics in the sediment transfer system.

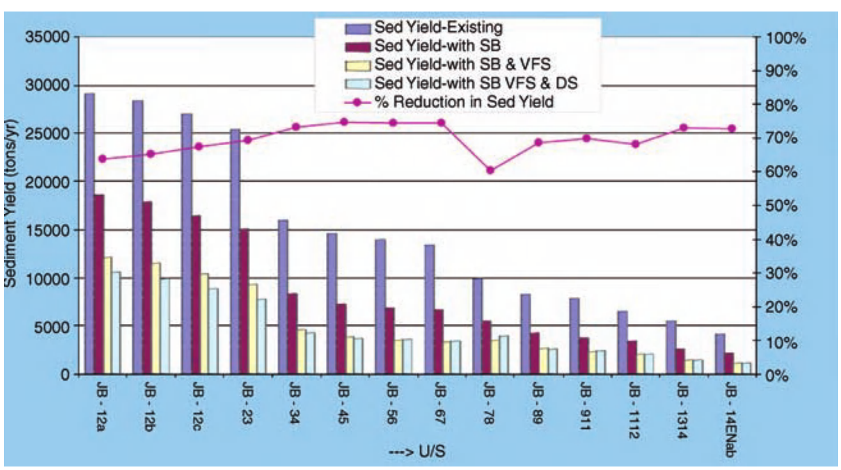


Fig. 5.9 Annualized sediment yields in Judy's Branch predicted by SIAM (Sediment Impact Analysis Method) for existing conditions and following implementation of a range of sediment source control measures. DS, drop structures; SB, small sediment basins; VFS, vegetative filter strips.

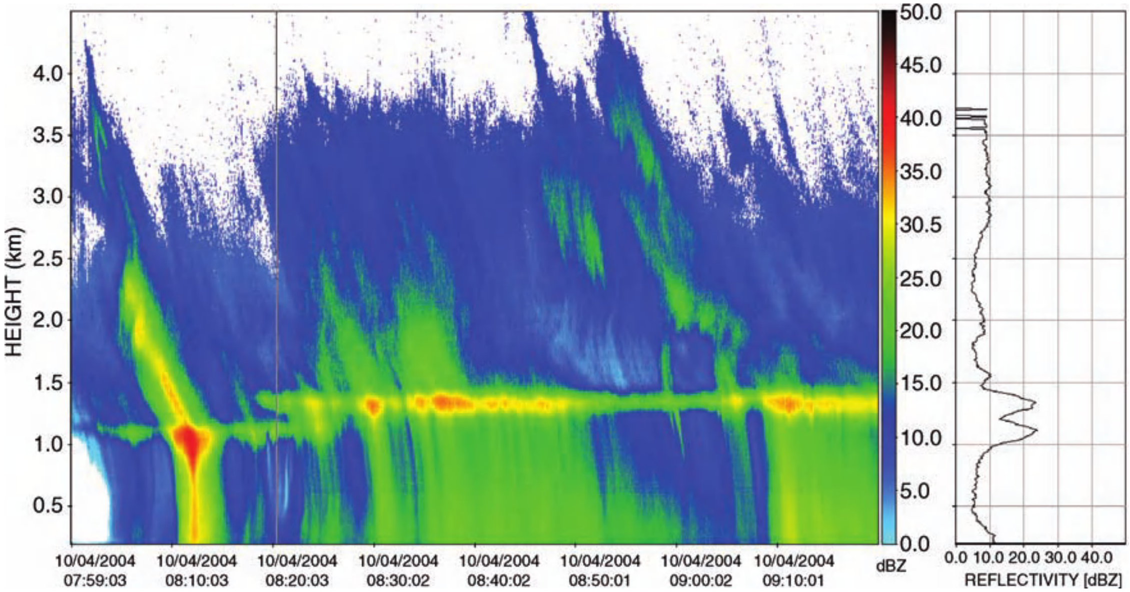


Fig. 7.1 Variation of the vertical reflectivity profile of precipitation. The data were obtained with a Vertically Pointing Radar (VPR) (see Cluckie *et al.* 2000).

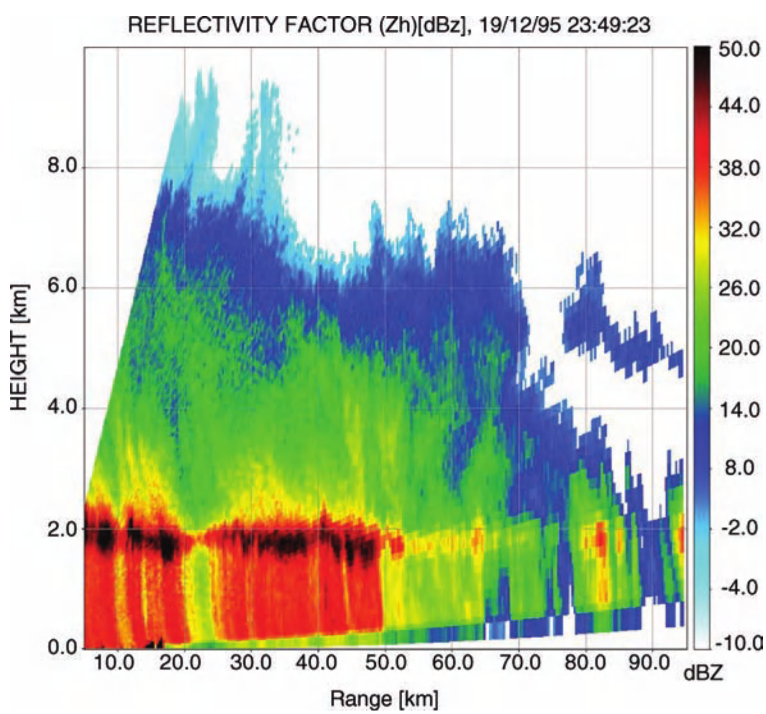


Fig. 8.4 A typical RHI scan for a stratiform event.

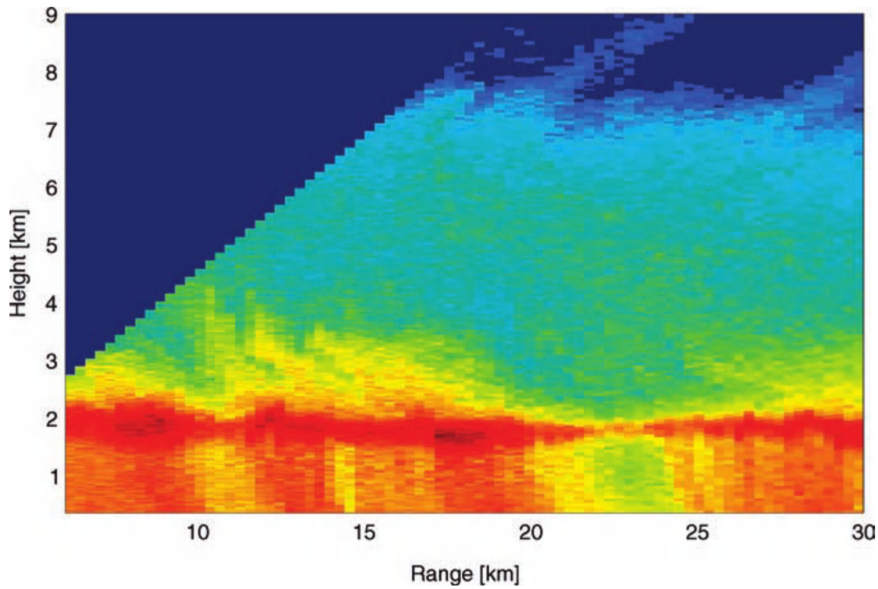


Fig. 8.5 RHI scan from the Chilbolton radar dataset.

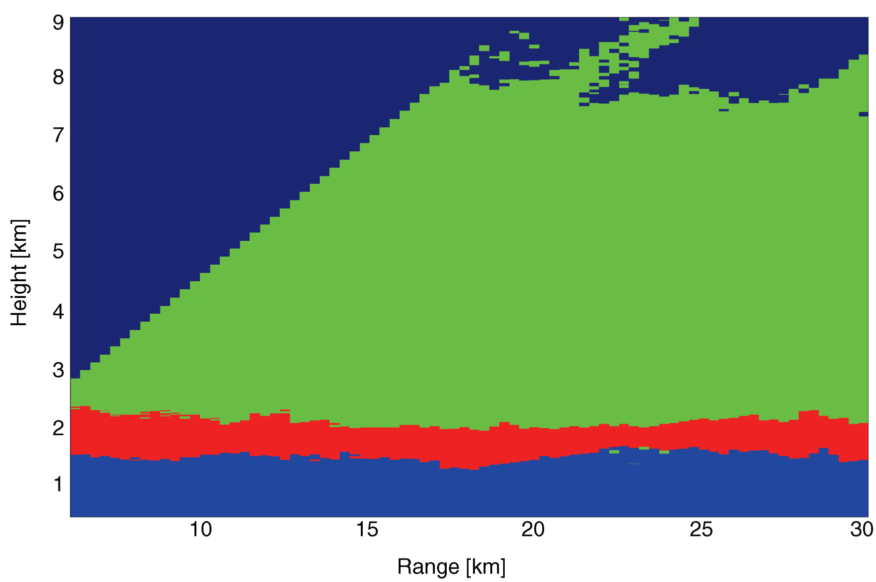


Fig. 8.6 Classification of scan in Figure 8.5 using the LID3 algorithm. Light blue indicates rain, green indicates snow, and red indicates bright band.

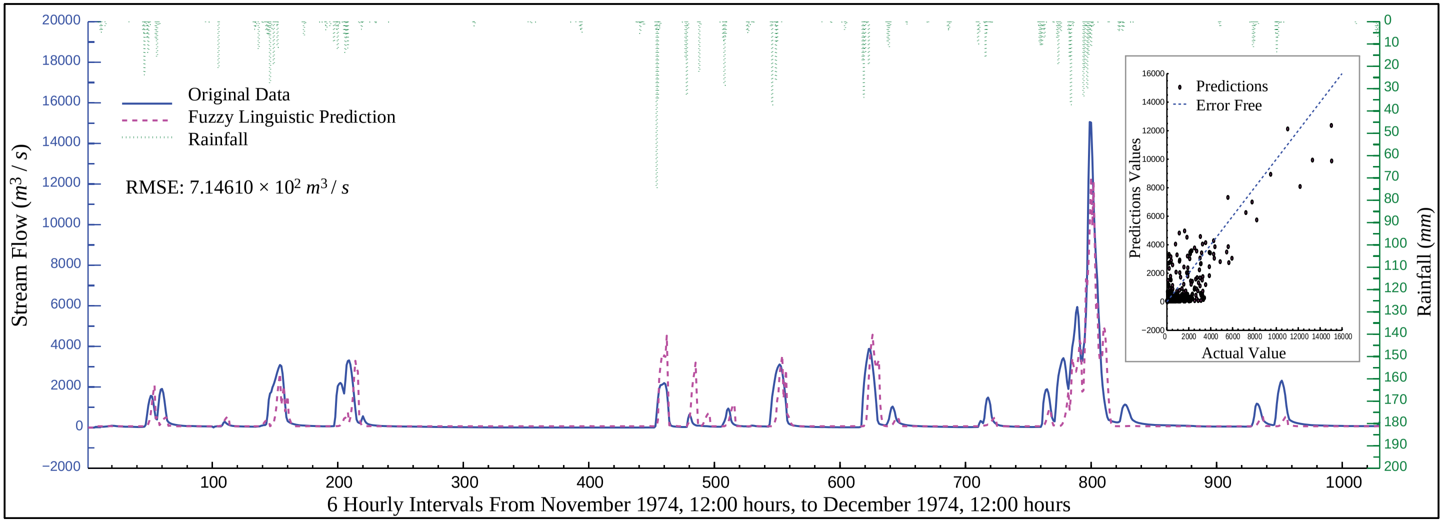


Fig. 8.7 Plots comparing actual flow against predicted flow applying the Fuzzy Bayesian prediction model (Randon *et al.* 2004). With kind permission from Springer Science & Business Media.

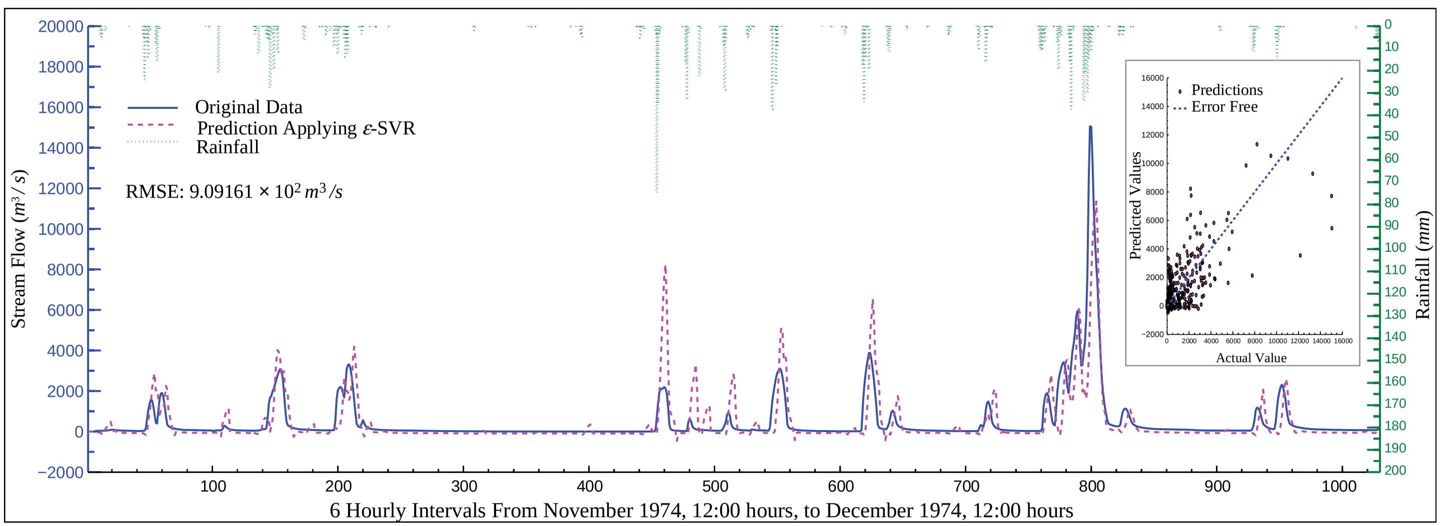


Fig. 8.8 Plots comparing actual flow against predicted flow applying an ε – SVR (Randon *et al.* 2004).

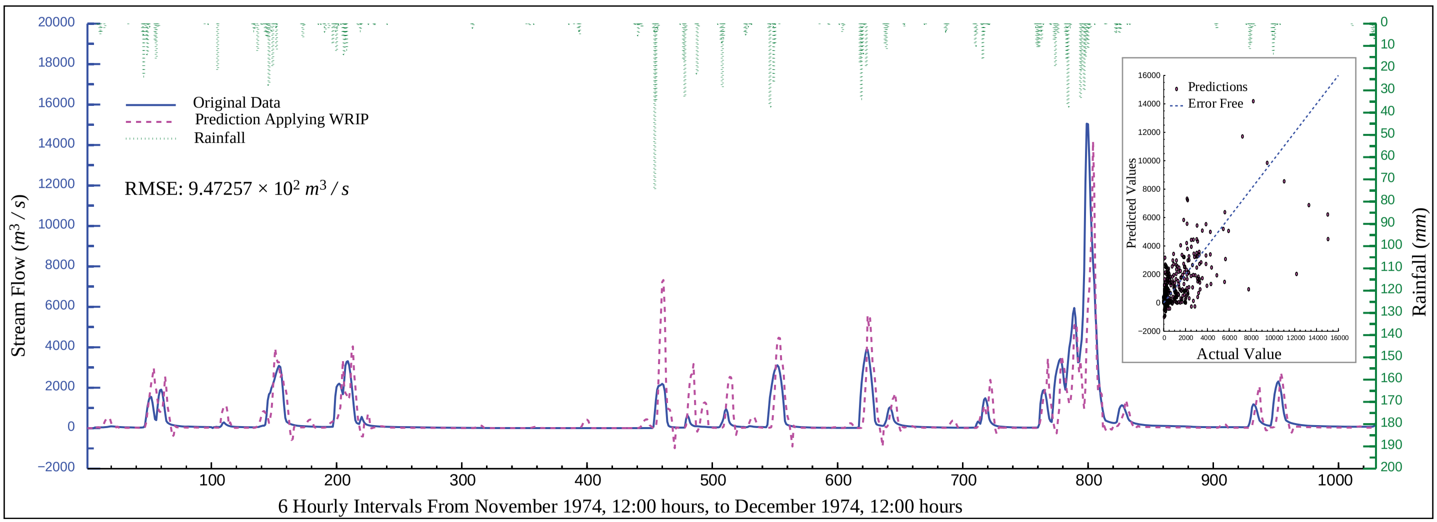
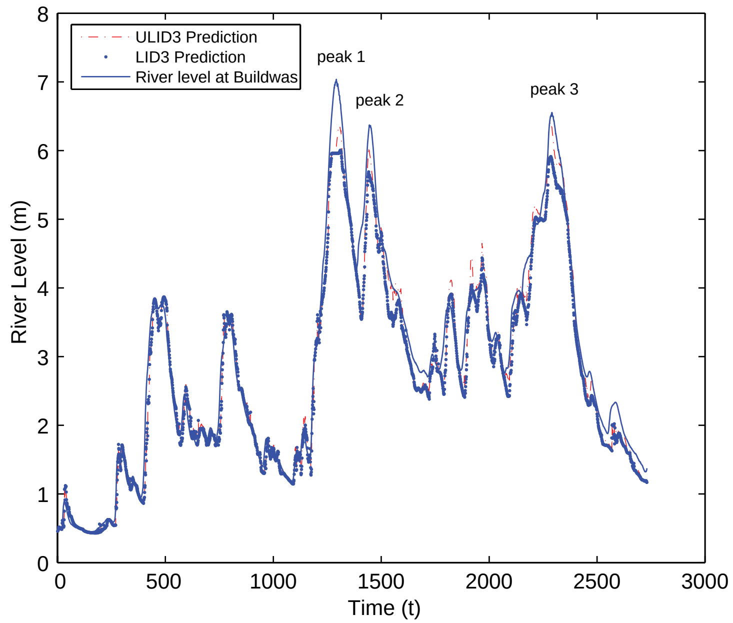
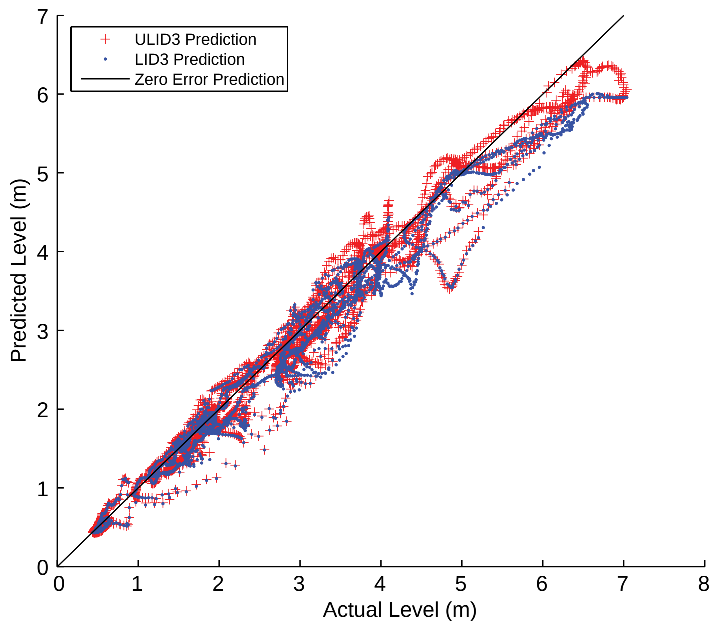


Fig. 8.9 Plots comparing actual flow against predicted flow applying the WRIP system.

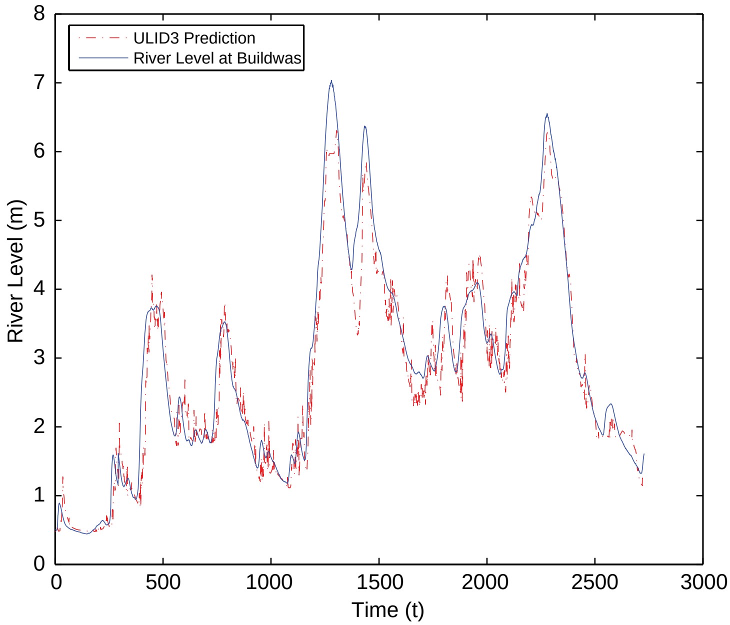


(a) Prediction of the River Level at Buildwas at $t + 24$ *hrs*

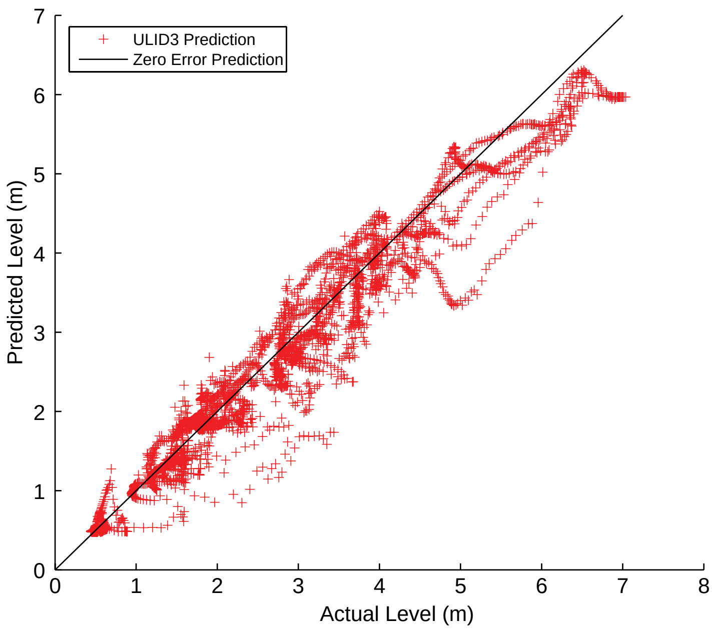


(b) Prediction of the River Level at Buildwas at $t + 24$ *hrs*

Fig. 8.10 ULID3 predictions of the river level at Buildwas at 24 hours ahead. (a) Predicted river level on the test data for both LID3 and ULID3 shown as a time series, together with the actual measurement values. (b) Scatter plot of predicted against actual values for both algorithms.

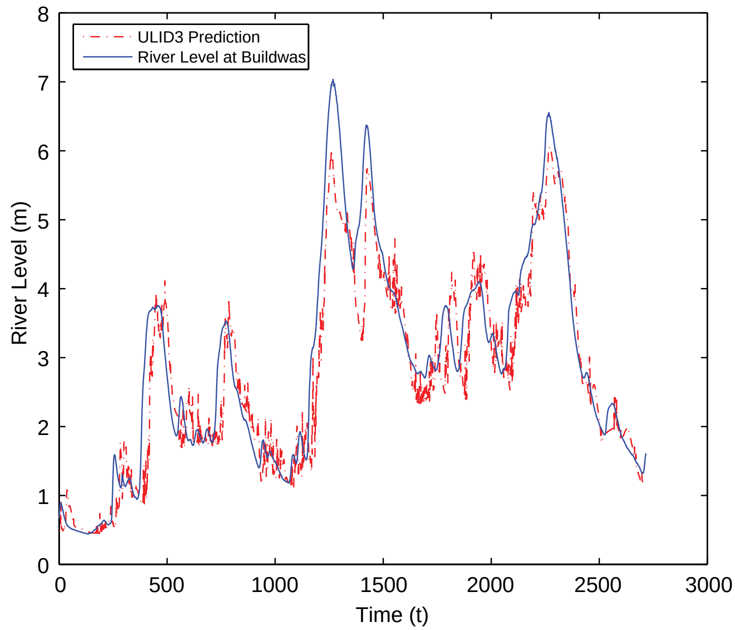


(a) Prediction of the River Level at Buildwas at $t + 36$ *hrs*

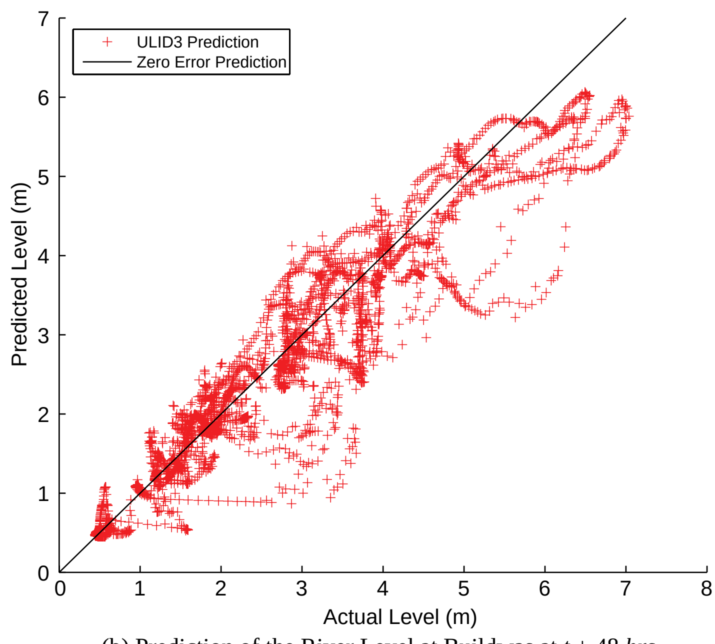


(b) Prediction of the River Level at Buildwas at $t + 36$ *hrs*

Fig. 8.11 ULID3 predictions of the river level at Buildwas at 36 hours ahead. (a) Predicted river level on the test data for ULID3 shown as a time series, together with the actual measurement values. (b) Scatter plot of predicted against actual values.



(a) Prediction of the River Level at Buildwas at $t + 48$ *hrs*



(b) Prediction of the River Level at Buildwas at $t + 48$ *hrs*

Fig. 8.12 ULID3 predictions of the river level at Buildwas at 48 hours ahead. (a) Predicted river level on the test data for ULID3 shown as a time series, together with the actual measurement values. (b) Scatter plot of predicted against actual values.

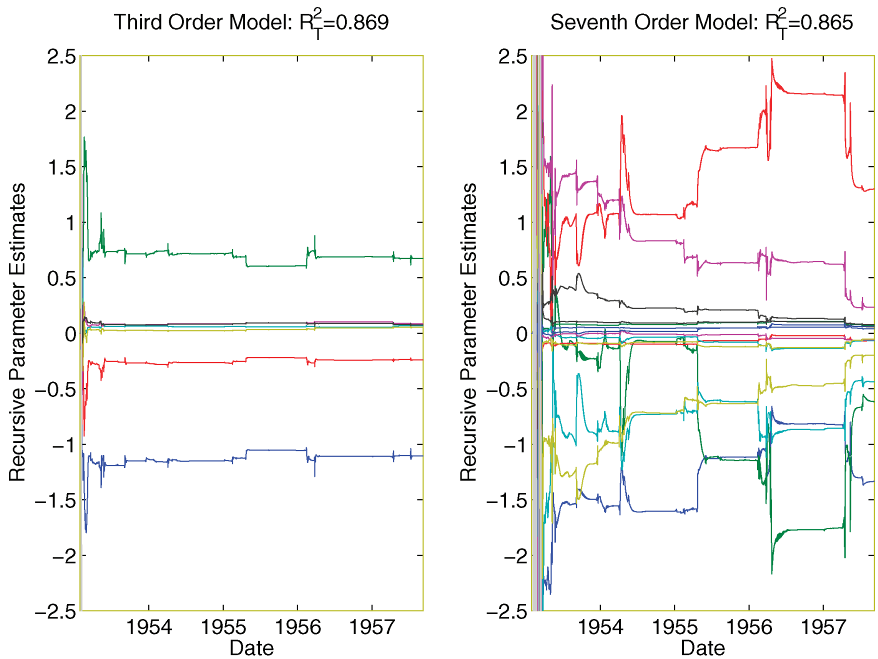


Fig. 9.4 Comparison of recursive estimates obtained for an identifiable third-order model (left panel) and a poorly identifiable seventh-order model (right panel).

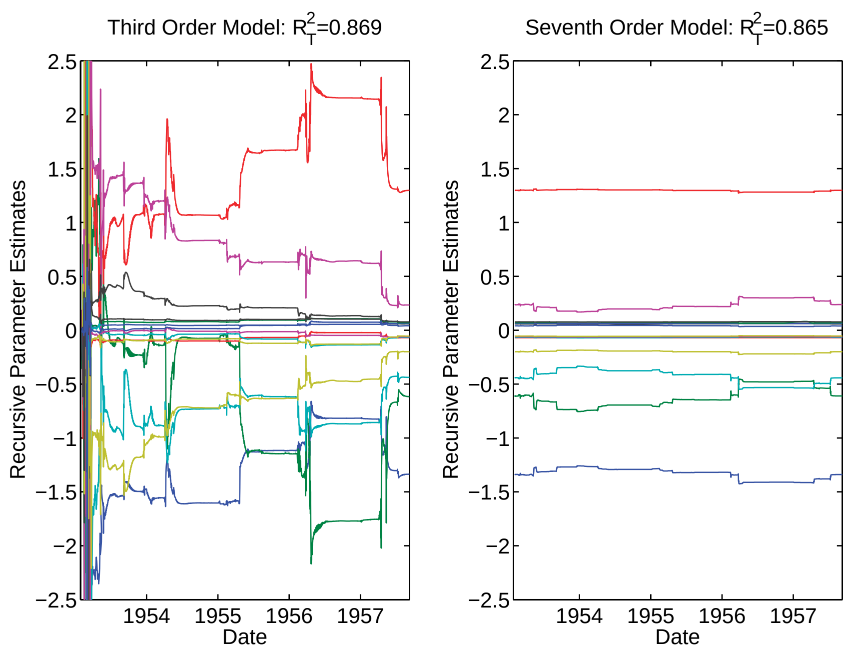


Fig. 9.7 The effect of imposing Bayesian prior constraints on the seventh-order model: no constraint with diffuse prior (left panel); tight constraint with Bayesian prior (right panel).

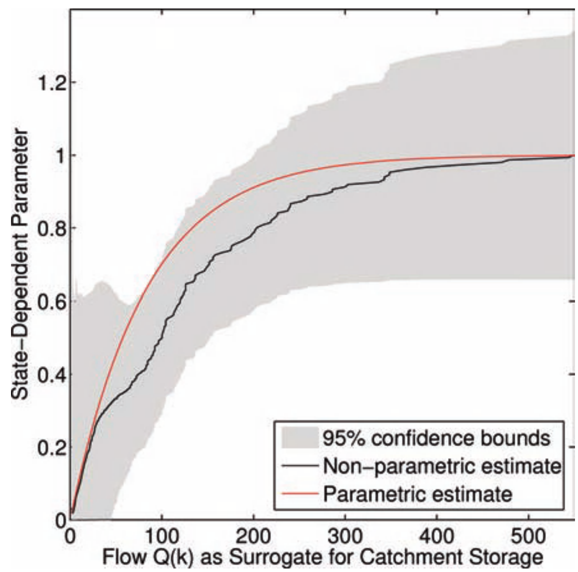


Fig. 9.10 Nonparametric (black line) and parametric (red line) estimates of the estimated State-Dependent Parameter (SDP) effective rainfall nonlinearity.

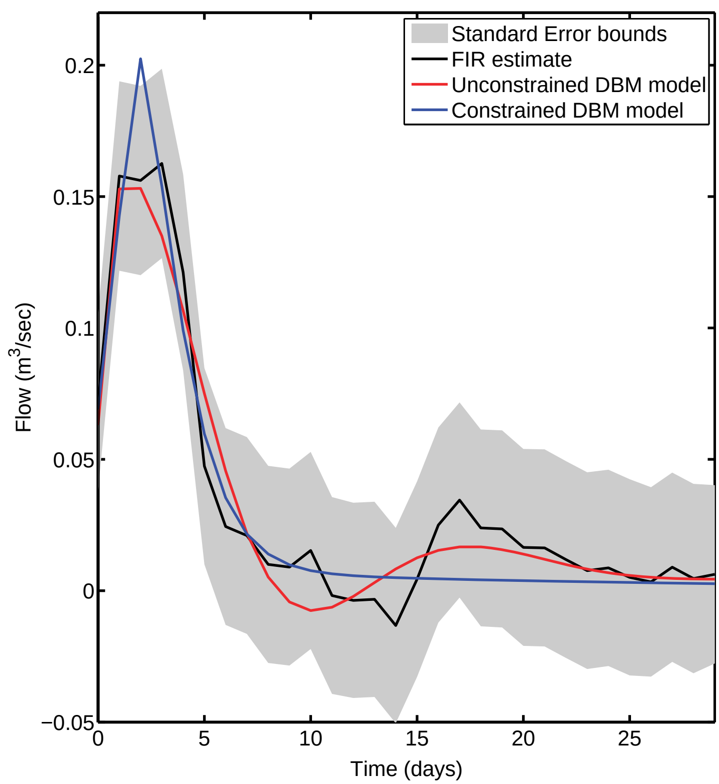


Fig. 9.11 Comparison of various unit hydrographs (impulse responses).

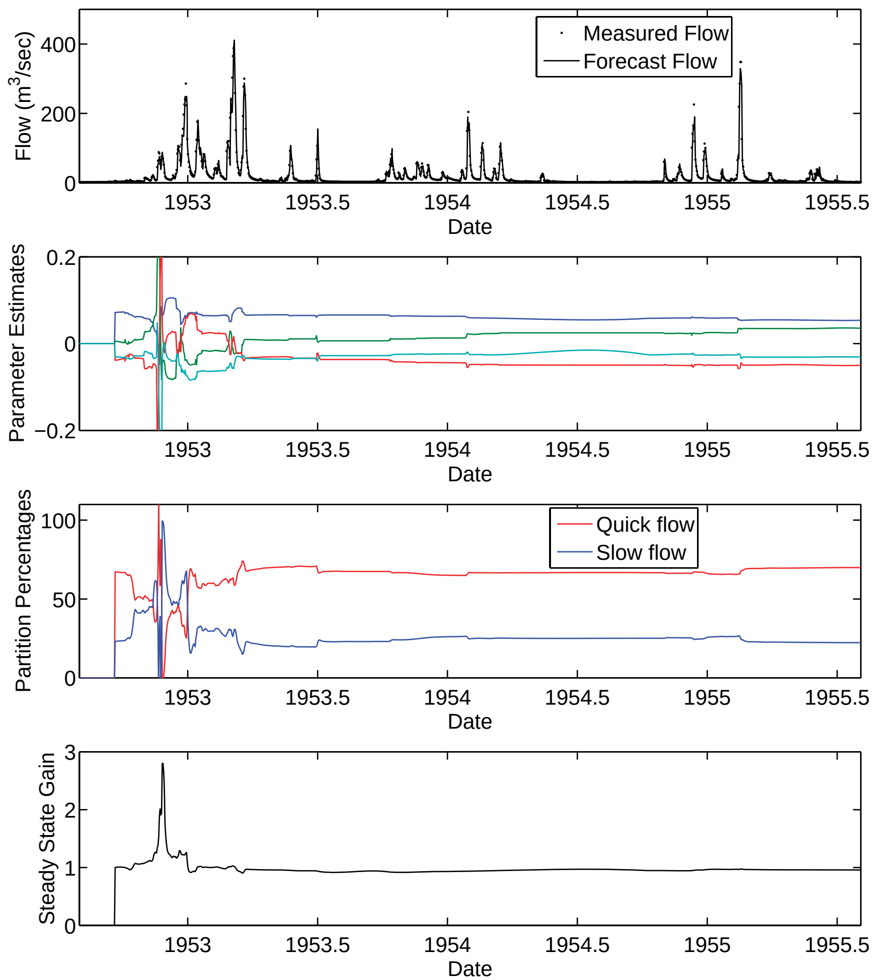


Fig. 9.12 Leaf River example: three years of real-time updating following initiation after 50 days: measured and forecast flow (upper panel); recursive estimates of model parameters (upper middle panel) and partition percentages (lower middle panel); and overall steady state gain (lower panel).

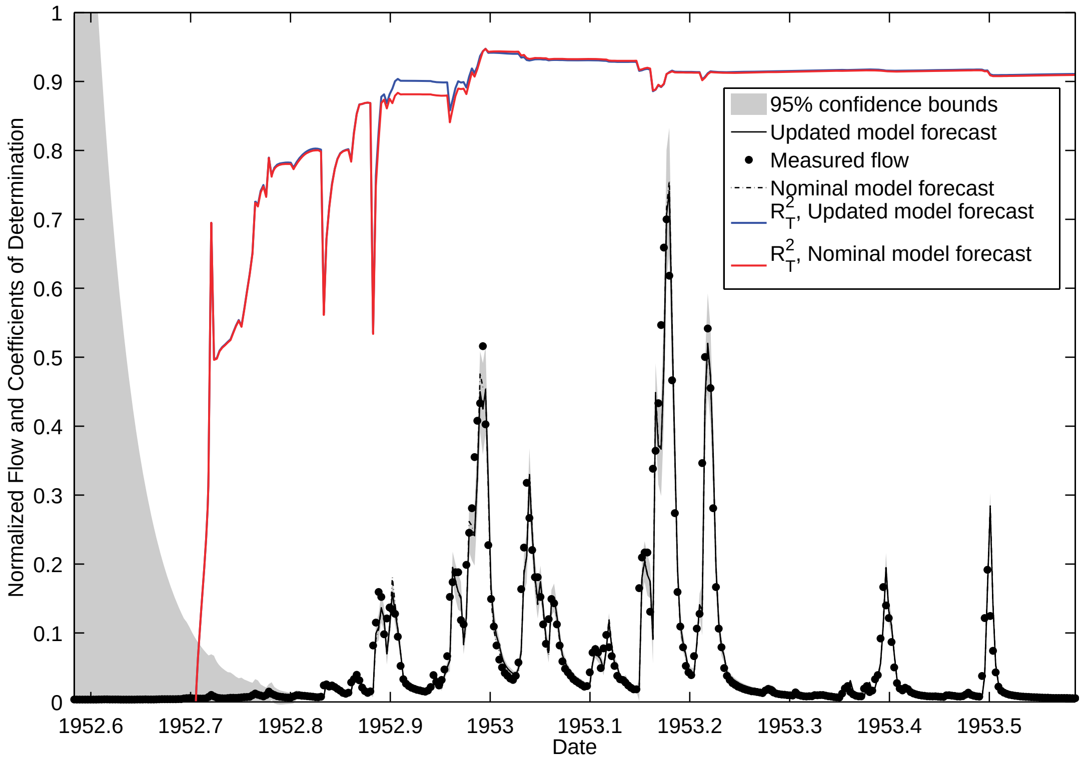


Fig. 9.13 Leaf River example: more detailed view of the real-time updated forecasting over the first year showing estimated 95% confidence bounds and running mean R_T^2 values for updated and fixed model forecasts, based on the innovation errors.

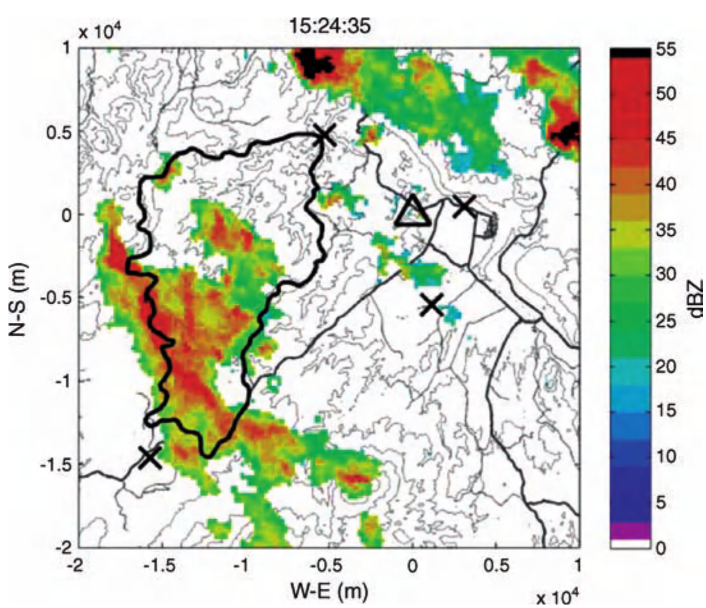


Fig. 10.3 Radar imagery of shower passing over a small catchment (outline solid black). The raingauge network was unable to sample spatial variability of the hydrometeor in this event. The radar is located at 0,0 (Δ). Terrain contours and roads have also been shown.

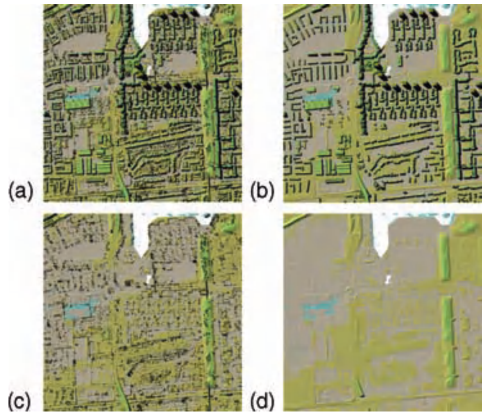


Fig. 11.2 Example of the Environment Agency's hybrid filtering. (a) Digital Surface Model; (b) Digital Terrain Model (DTM) with buildings; (c) DTM with vegetation; and (d) DTM without bridges. Reproduced with permission of Geomatrics Group, Environment Agency.

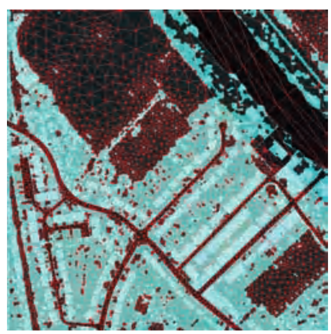


Fig. 11.3 Mesh constructed over vegetated urban area (red = mesh, blue = building/taller vegetation heights; a river is present in the northeast corner). After Mason *et al*. (1996).

Fig. 12.9 Modelling of urban inundation at high resolution.

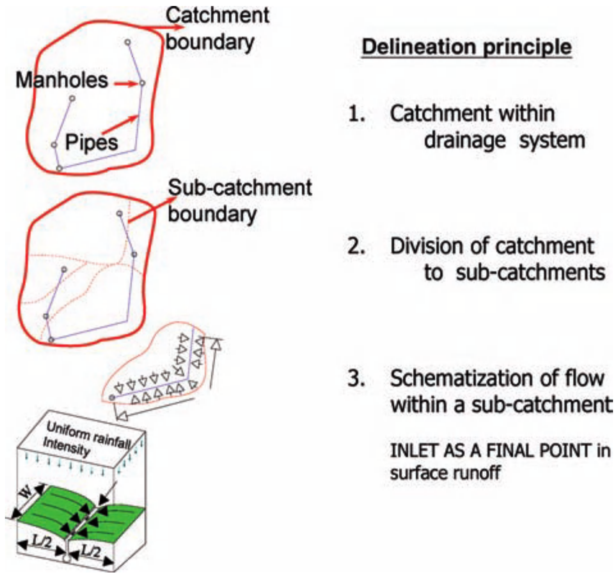


Fig. 13.7 Subcatchment delineation.
L = catchment length, W = catchment width.

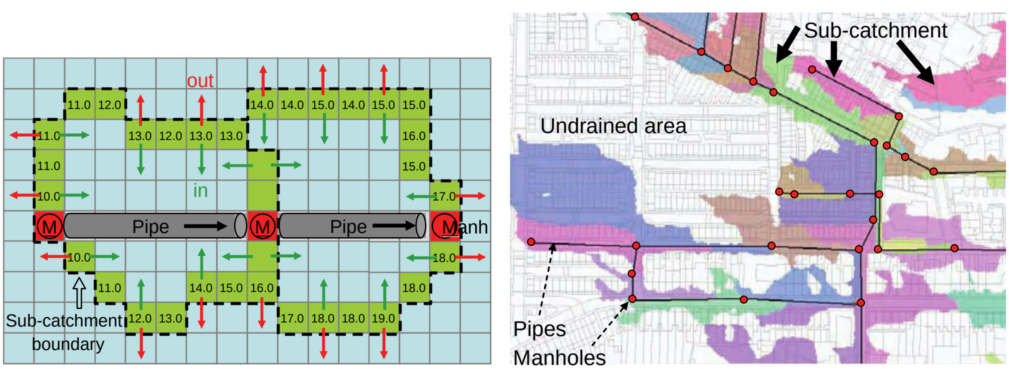


Fig. 13.8 Link based approach to determine sub-catchment delineation.

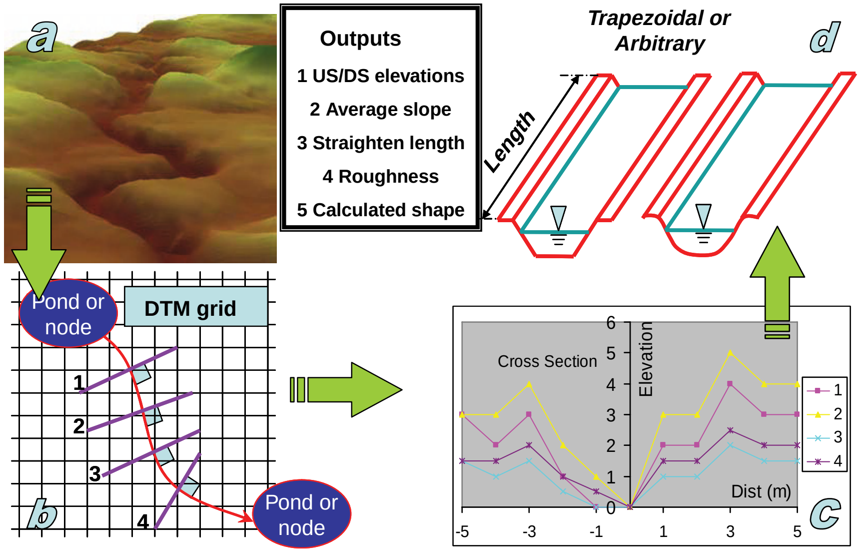


Fig. 13.10 Estimation of pathways geometry. (a) A 3D digital terrain model (DTM) showing identified flow path (U/S upstream, D/S downstream). (b) A number of cross-sectional lines drawn perpendicularly to the path. (c) The arbitrary shapes of cross-sections plotted as found from the DTM. (d) Averaged output with two choices, trapezoidal or actual shapes.

Fig. 13.11 Ponds and flowpaths derived from the digital elevation model (DEM). Flowpaths in blue; location of ponds in yellow; surface water system in green; combined sewer system in red; node manholes as small black circles; gullies identified as dots by the kerb lines.

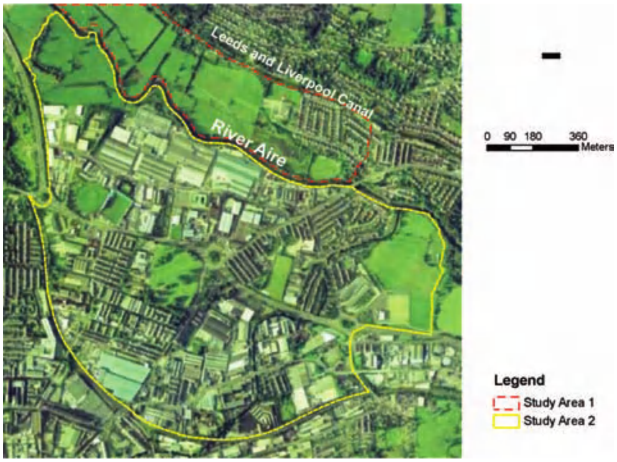


Fig. 13.18 Aerial photo of the study areas: Stockbridge in Keighley, West Yorkshire.

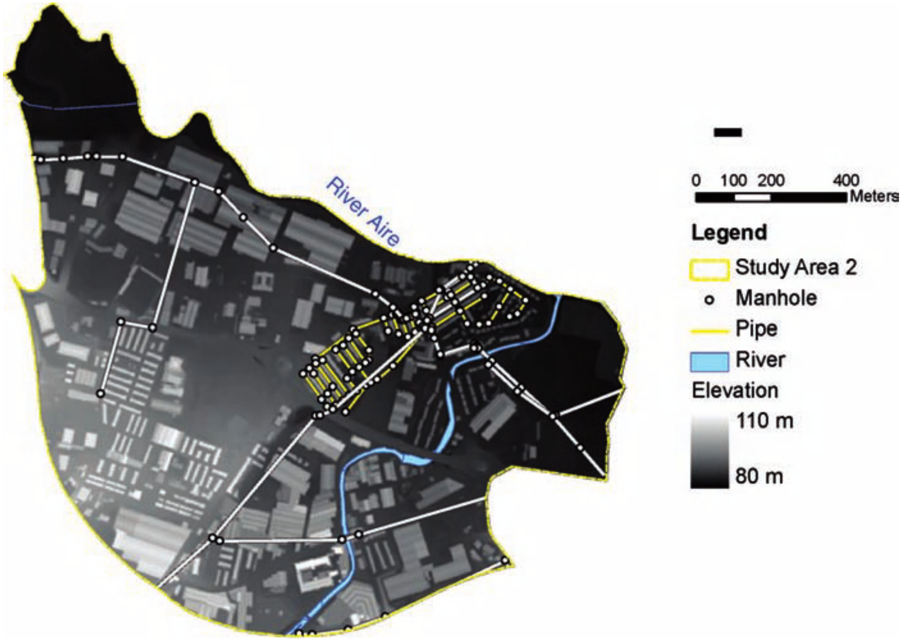


Fig. 13.19 The drainage system and the digital elevation model (DEM) of the study area.

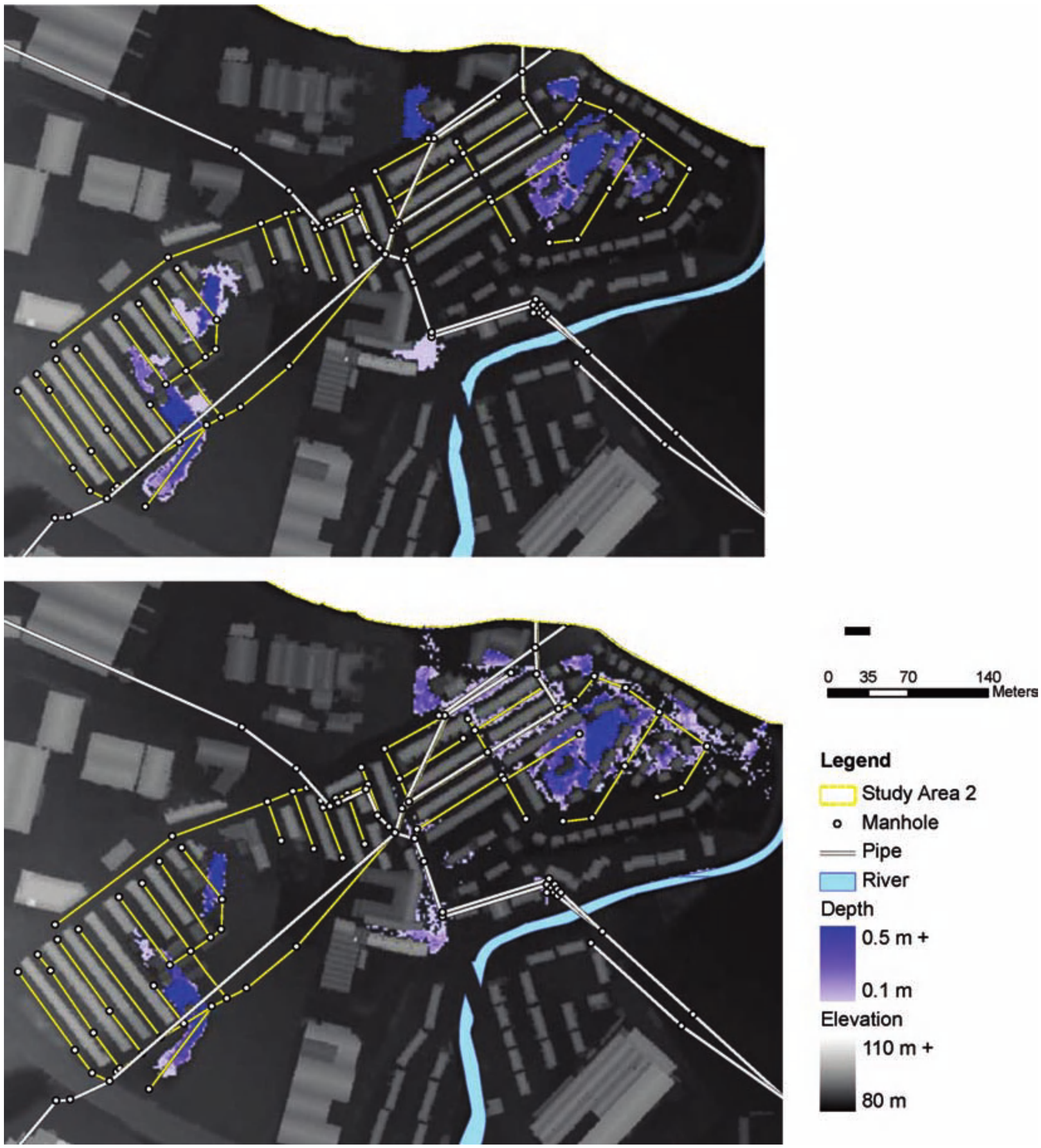


Fig. 13.20 Simulation results of 1D/1D (above) and 1D/2D (below) modelling.

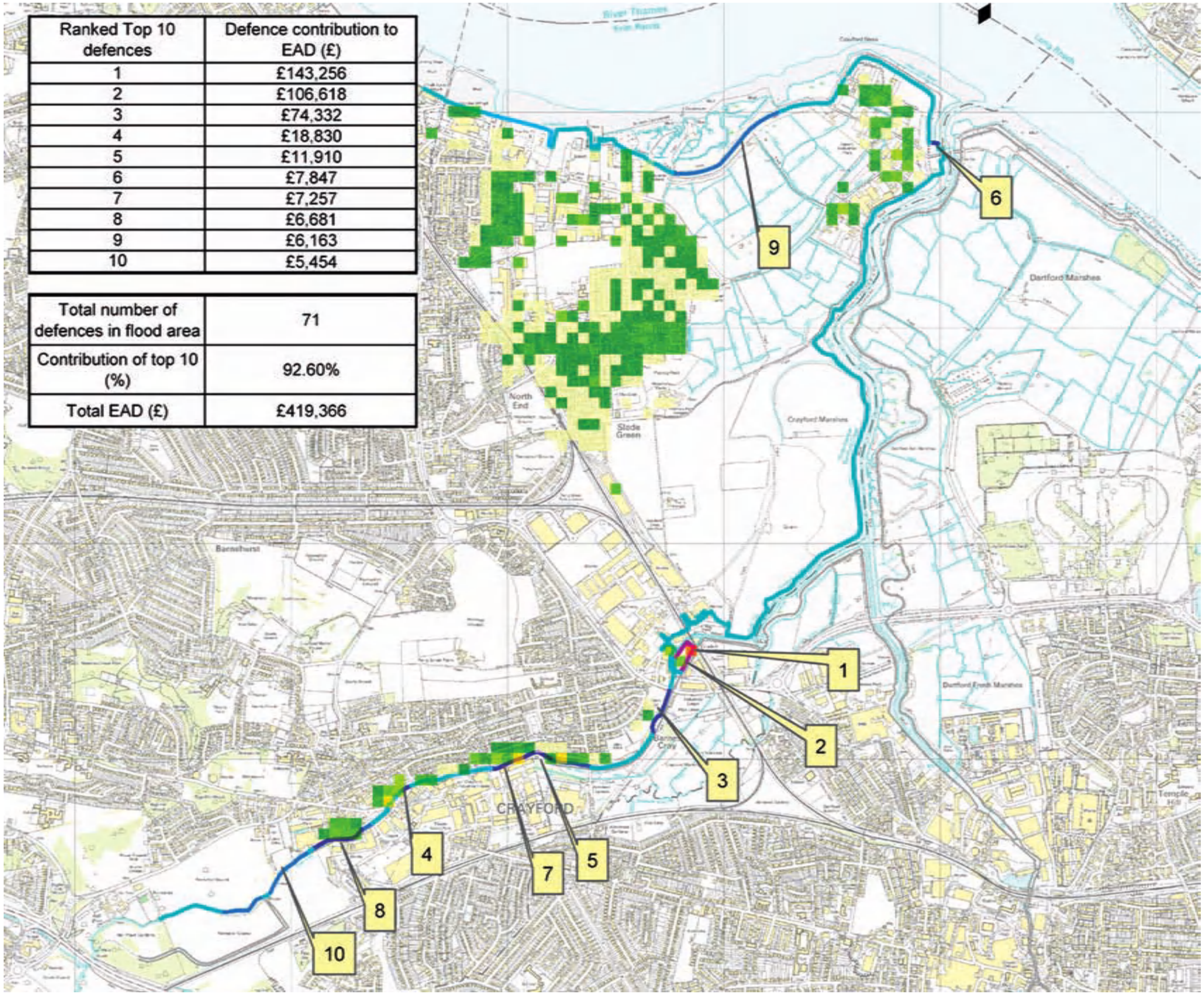


Fig. 15.2 The expected annual damage (EAD) attributed to individual defence assets within an asset system (as well as spatially within the floodplain).

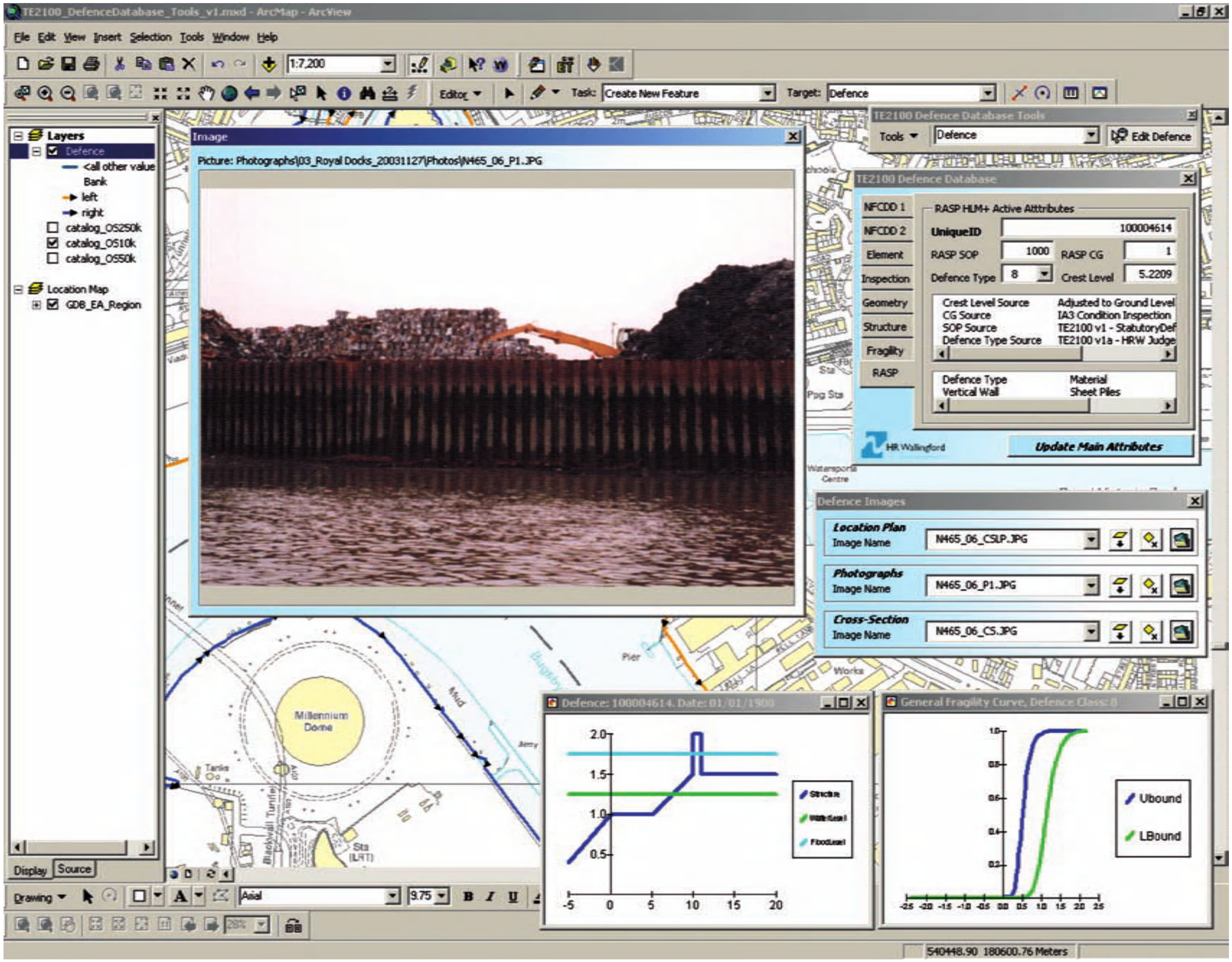


Fig. 15.5 A common asset database as used and shared across the Thames Estuary Planning for Flood Risk Management 2100 (Sayers *et al.* 2006).

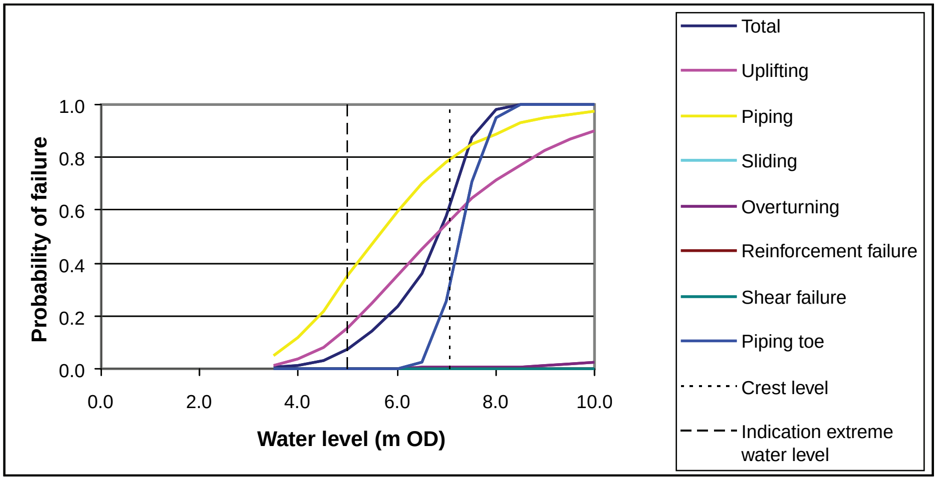


Fig. 15.6 A typical fragility curve based on the reliability analysis for a defence in the Thames Estuary.

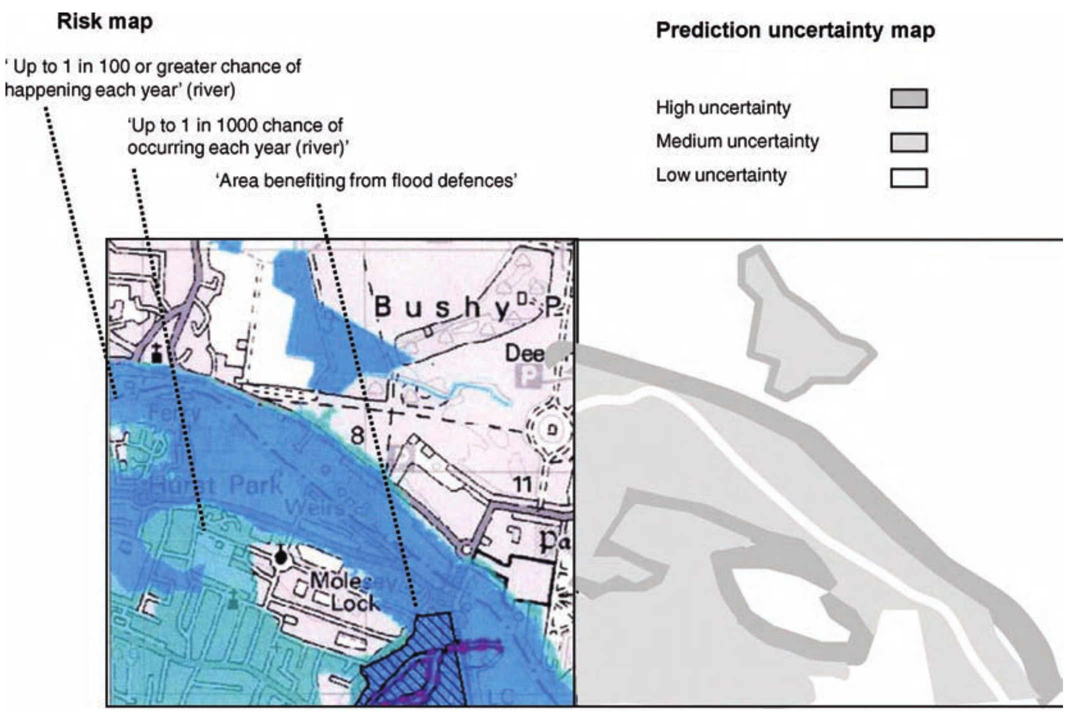


Fig. 19.1 Left: Indicative floodplain maps, as available on the UK's Environment Agency (EA) website. Right: Pattern of uncertainty associated with the risk estimation.

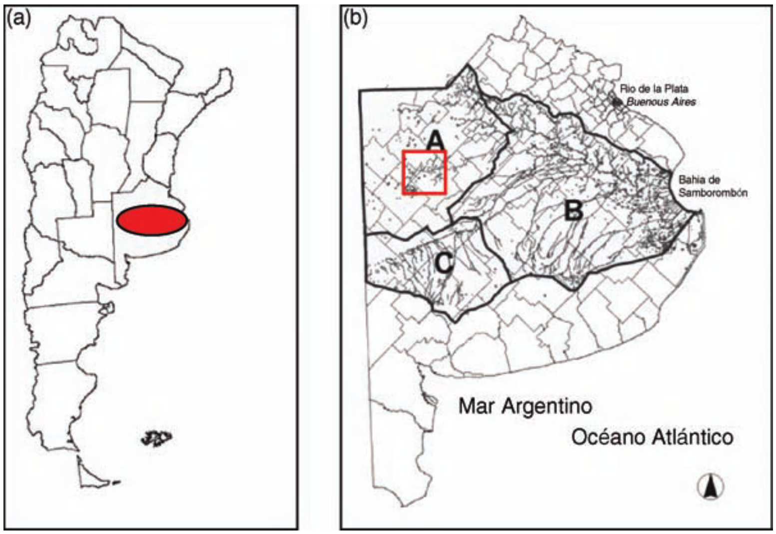


Fig. 22.1 (a) Location of Buenos Aires Province within Argentina. (b) Location of Río Salado Basin within Buenos Aires Province. A: Northwest Region; B: Salado, Vallimanca and Las Flores river valleys; C: South-Western Lake System. The red box shows the location of a dune field area within the Northwest Region (A).

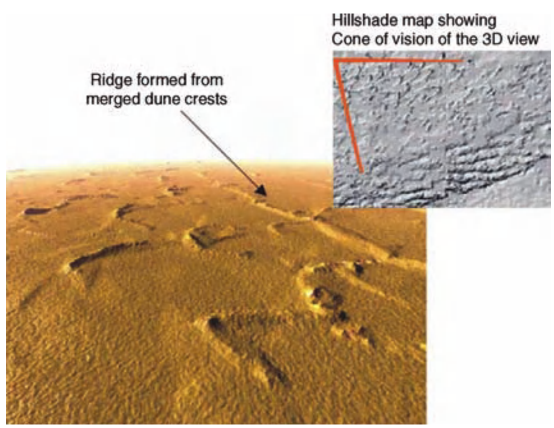


Fig. 22.2 Three-dimensional view of aeolian features in the Northwest of the Buenos Aires Province (red box in Fig. 22.1b).

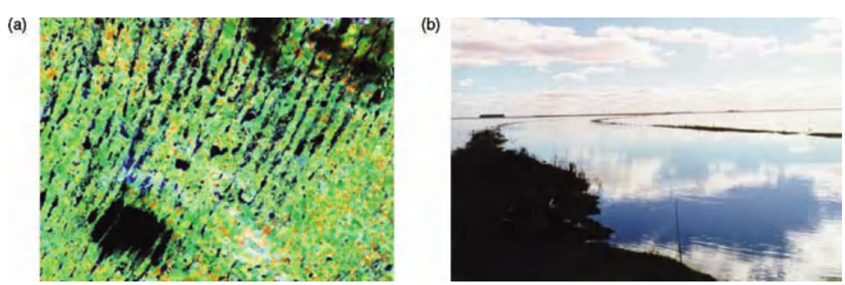


Fig. 22.3 (a) Satellite image showing flooding in the Northwest of Buenos Aires Province (Region A) – see flooding captured behind the dune crests shown in Fig. 22.2. (b) Flooding in the Río Salado (Region B).

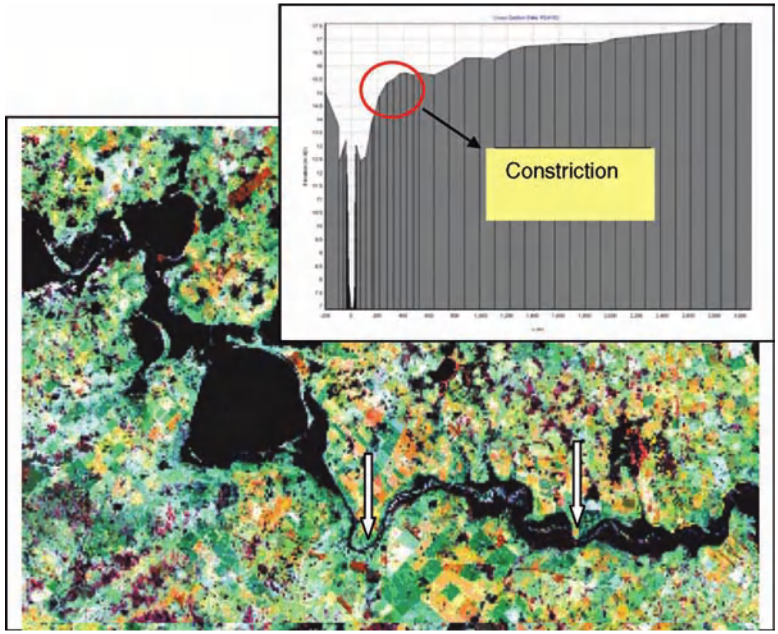


Fig. 22.6 Interaction between relict aeolian features and fluvial flooding in the Lower Salado.

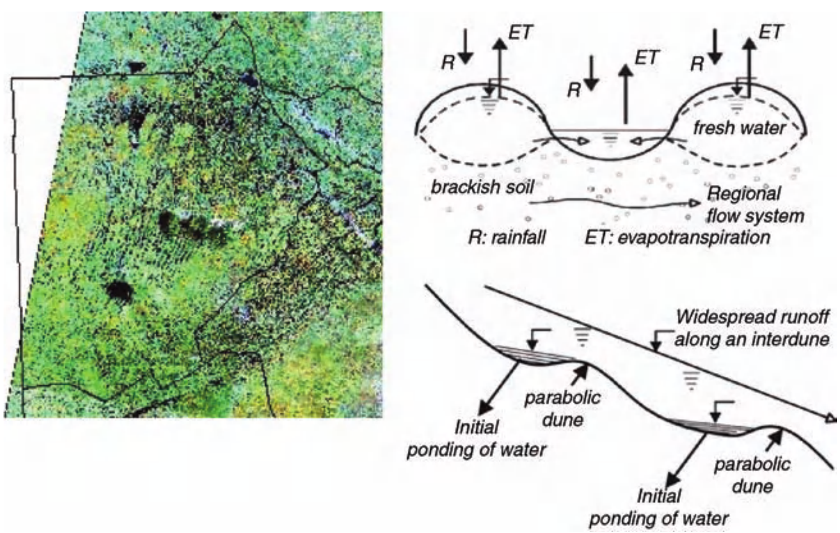


Fig. 22.7 Conceptual representation of groundwater-induced surface flooding in the relict dune field of the Northwest part of the Río Salado Basin (Region A).

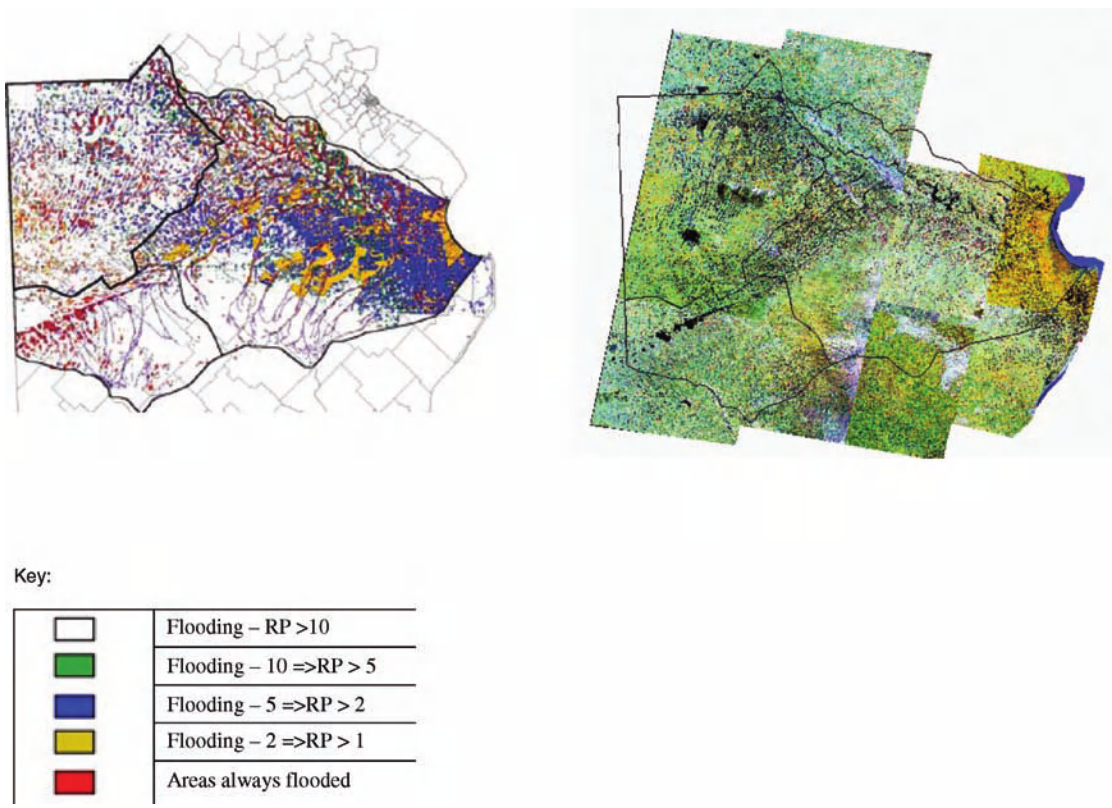


Fig. 22.10 Sample Flood Probability Map (FPM) produced using the regional flood model. RP, return period.

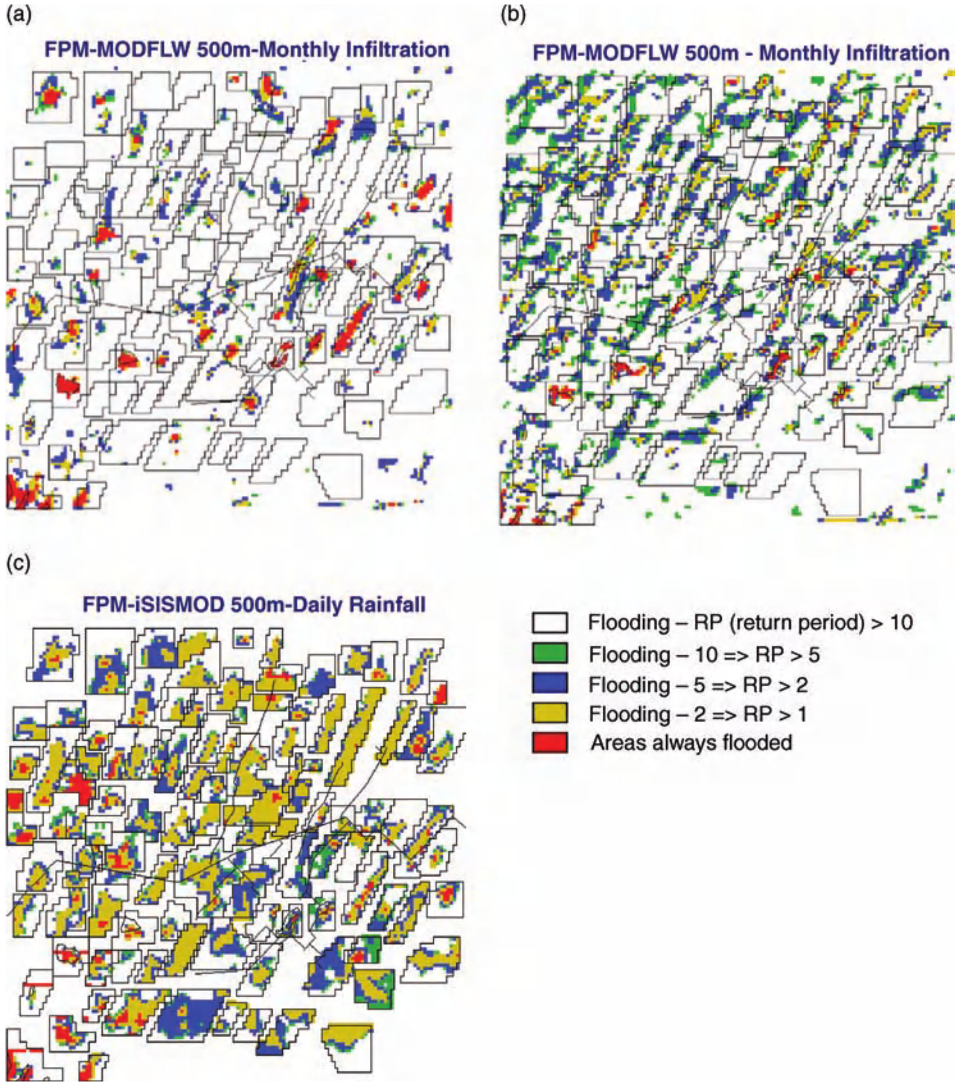


Fig. 22.12 Flood Probability Maps (FPMs). (a) MODFLOW – 5000 m – Monthly infiltration. (b) MODFLOW – 500 m – Monthly infiltration. (c) iSISMOD – 500 m – Daily rainfall.

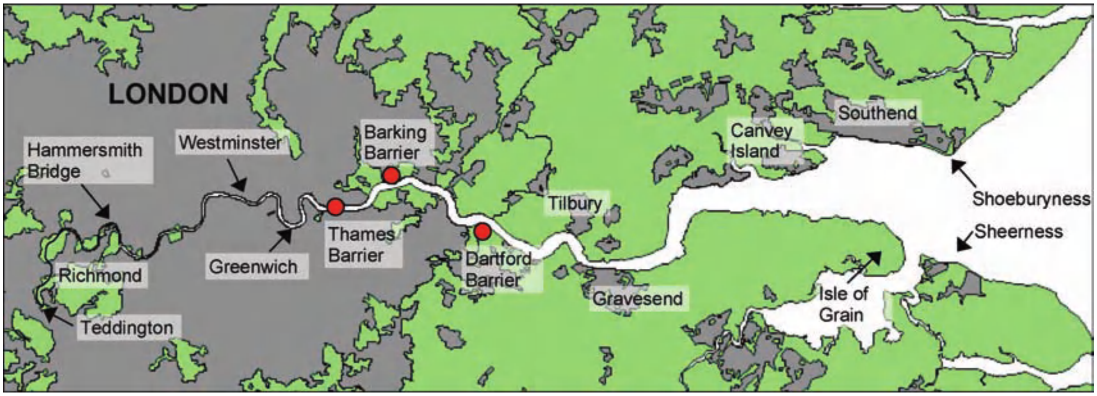


Fig. 23.1 Thames Estuary location plan.

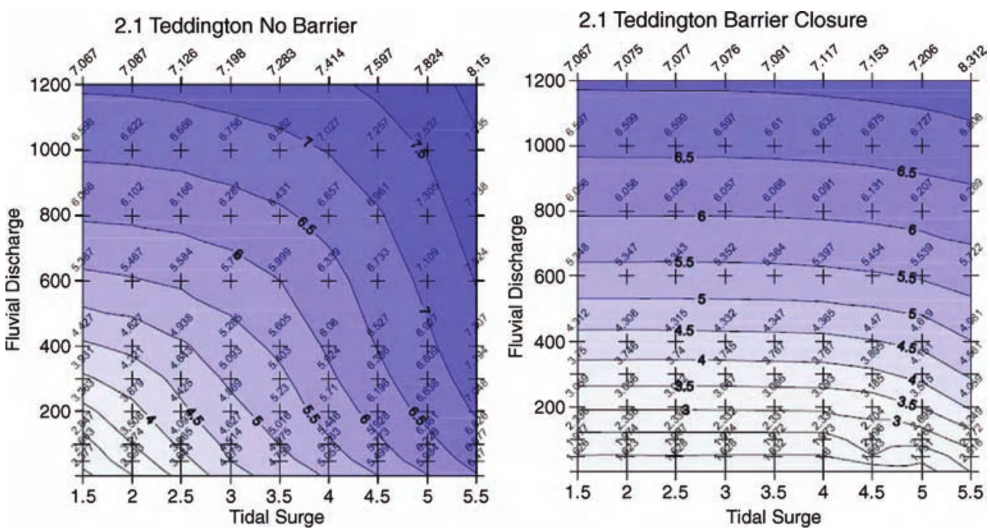


Fig. 23.3 Example structure function for Teddington (ISIS model node 2.1).

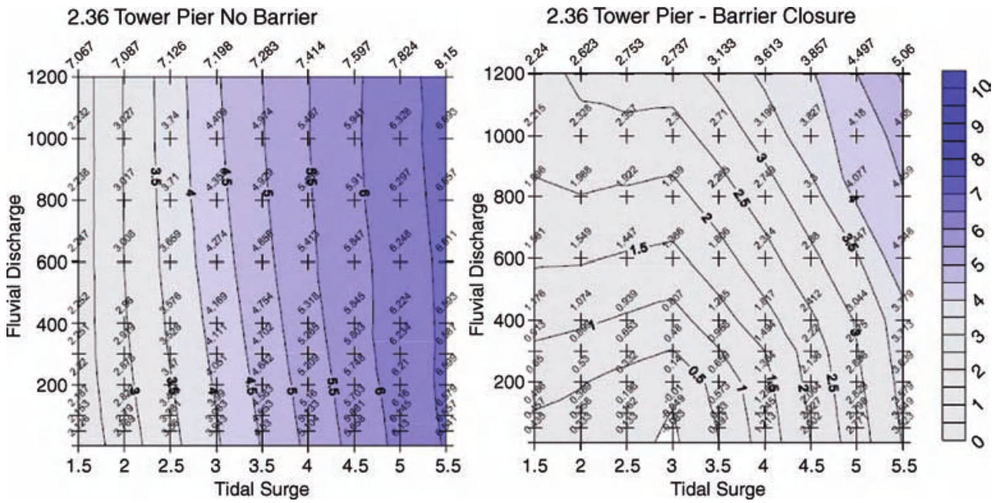


Fig. 23.4 Example structure function for Tower Pier (ISIS model node 2.36).

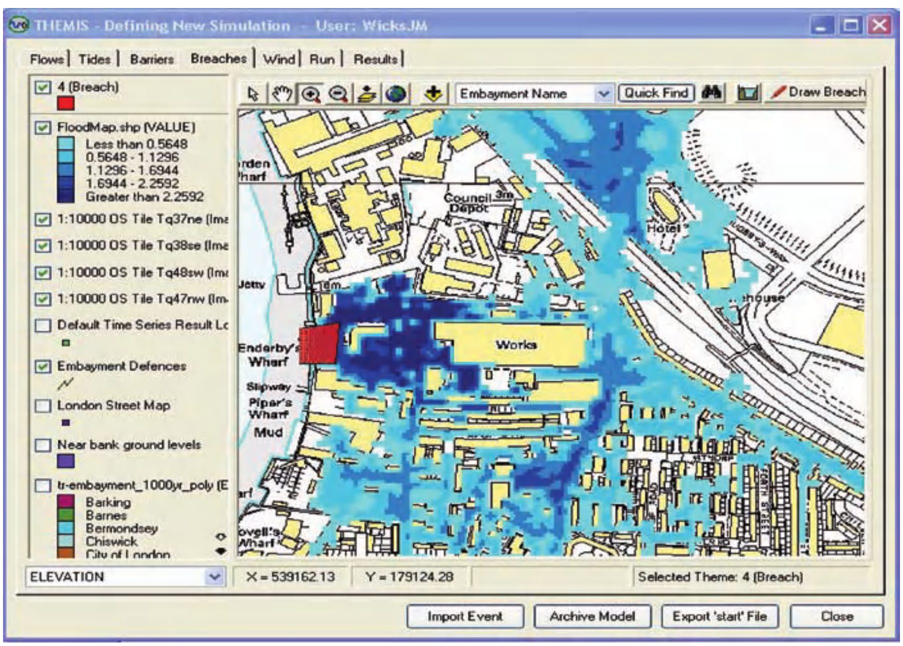


Fig. 23.5 Predicted flooding following a hypothetical breach at Enderby's Wharf.

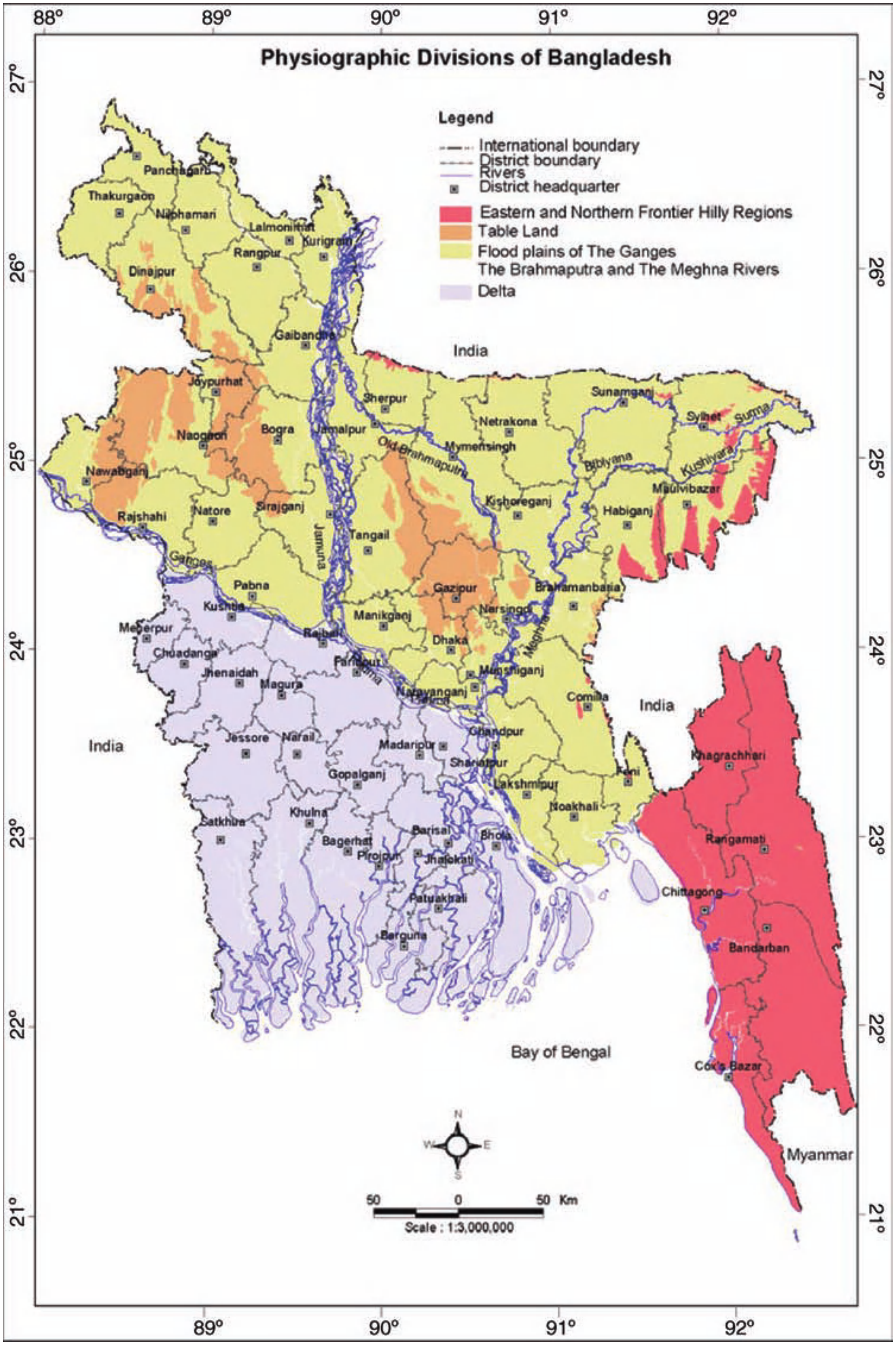


Fig. 24.1 Map of Bangladesh showing the physiographic features of the country.

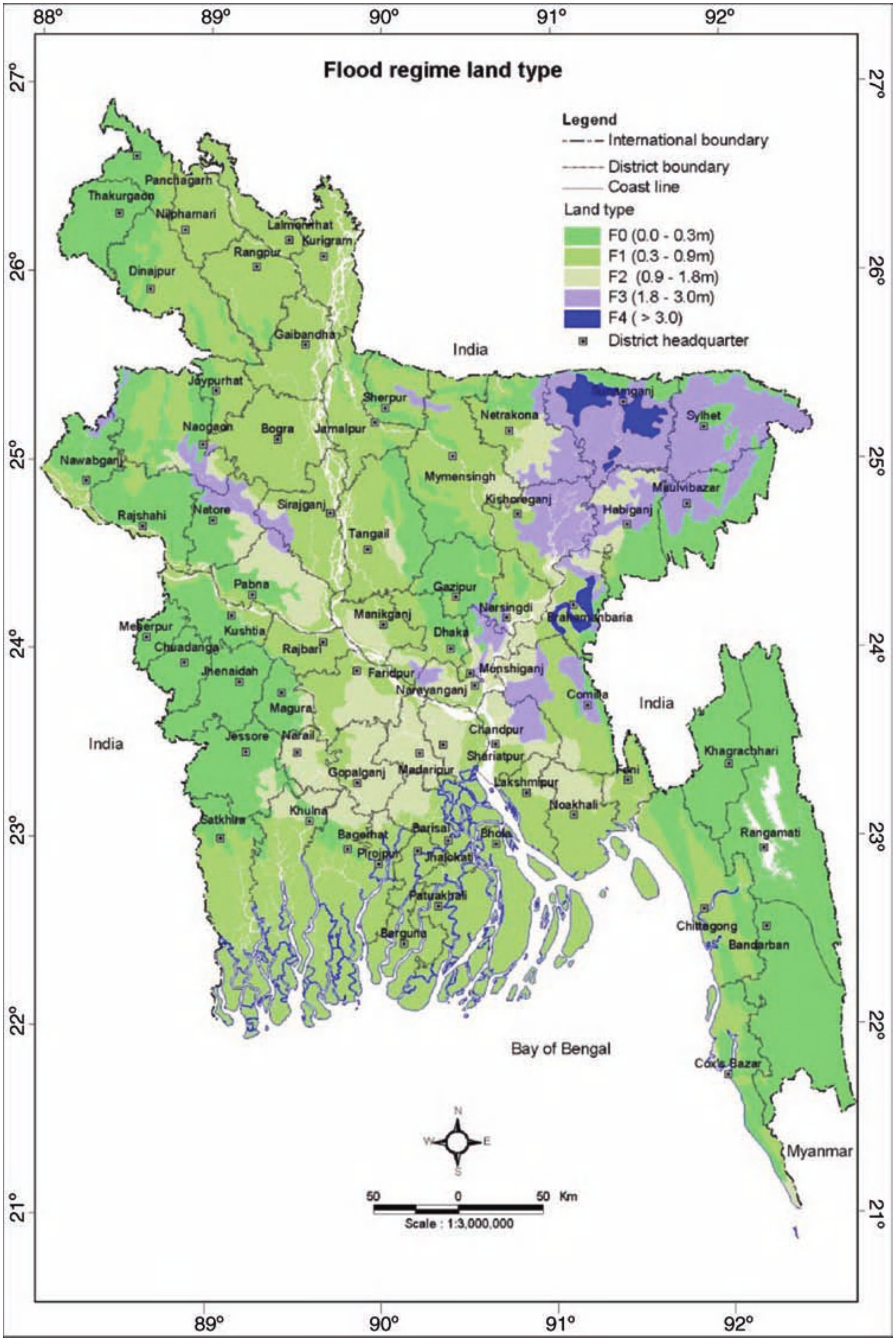


Fig. 24.2 Land type according to flood depth in Bangladesh.

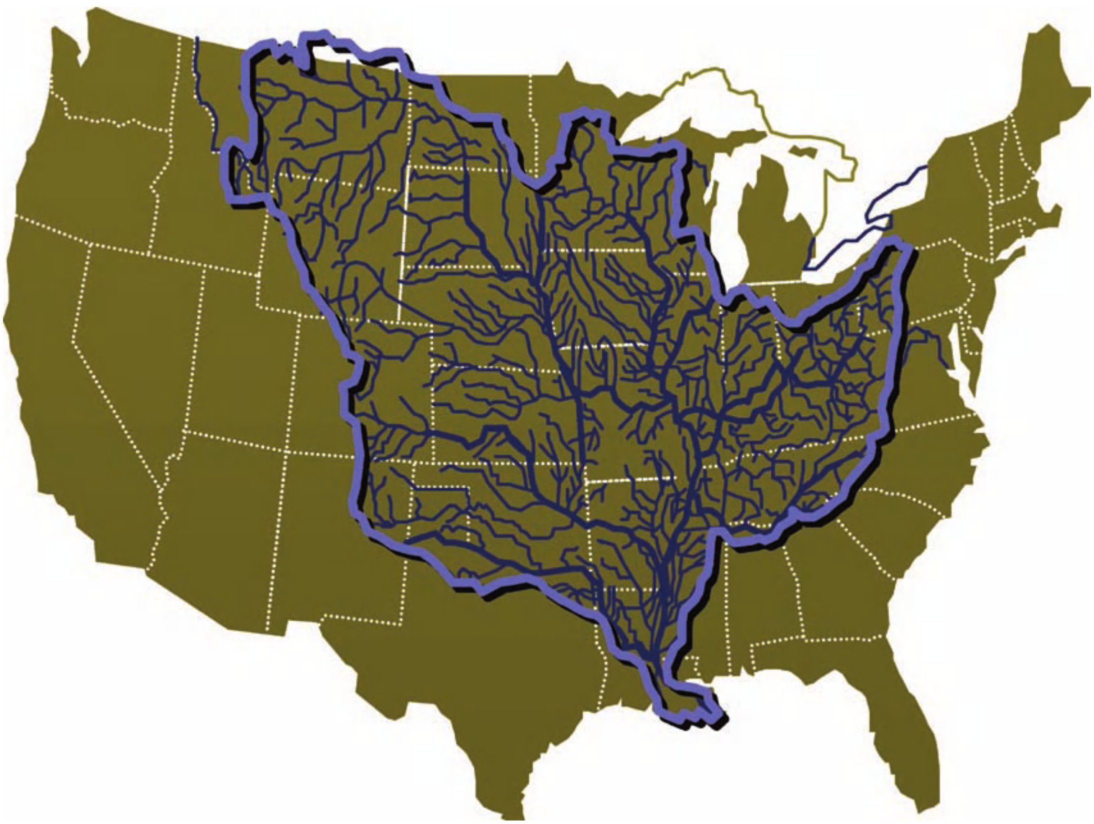


Fig. 25.1 The Mississippi River Basin.

Part 1
Introduction

Flood Risk Science and Management
Edited by Gareth Pender and Hazel Faulkner
 ISBN: 978-1-405-18657-5

1 Setting the Scene for Flood Risk Management

JIM W. HALL AND EDMUND C. PENNING-ROWSELL

The Changing Context of Modern Flood Risk Management

A major shift in approaches to the management of flooding is now underway in many countries worldwide. This shift has been stimulated by severe floods, for example on the Oder (Odra; 1997), Yangtze (1998), Elbe (Labe; 2002), Rhône (2003), in New Orleans (2005), on the Danube (2006) and in the UK (2000, 2007 and 2009). Also important has been a recognition of the relentless upward global trend in vulnerability to flooding and hence losses (Munich Re Group 2007), as well as threats from the potential impacts of climate change on flood frequency. In this context this chapter examines the main characteristics of the emerging approach to flood risk management, as a prelude to the more detailed exploration of methods and models that follows in this volume.

Whilst recent floods have been a stimulus for changing flood risk management policy and practice in the UK (Johnson 2005; Penning-Rowsell 2006), the notion of an integrated risk-based approach to flood management is in fact well established (National Academy of Engineering 2000; National Research Council 2000; Sayers *et al.* 2002; Hall *et al.* 2003c). Methods for probabilistic risk analysis have been used for some years in the narrower context of flood defence engineering (CUR/TAW 1990; Vrijling 1993; USACE 1996; Goldman 1997). Indeed the notion of risk-based optimization of the costs and benefits of flood defence was laid out in van Dantzig's (1956) seminal analysis.

However, modern flood risk management no longer relies solely upon engineered flood defence structures, such as dikes, channel improvement works and barriers. It also considers a host of other measures that may be used to reduce the severity of flooding (e.g. land use changes in upstream catchments) or reduce the consequence of flooding when it does occur, by reducing either exposure (White and Richards 2007; Richards 2008) or vulnerability (Tapsell 2002). The criteria for the assessment of flood risk management options are now seldom solely economic (Penning-Rowsell *et al.* 2005; Johnson 2007a), but involve considerations of public safety (Jonkman and Penning-Rowsell 2008), equity (Johnson 2007b) and the environment (Green 2004). Furthermore, an increasing recognition of non-stationarity (Milly *et al.* 2008) means that flood risk management involves explicit consideration of the ways in which flood risk may change in future, due, for example, to climate change or the apparently inexorable process of floodplain development (Parker and Penning-Rowsell 2005). This leads to the notion of flood risk management being a continuous process of adaptive management rather than a 'one-off' activity (Hall *et al.* 2003c; Hutter and Schanze 2008).

The locus of power is also changing in many countries as governments seek more effective and efficient institutional arrangements. In the UK, as well as the devolved administrations in Wales and Scotland now taking somewhat different

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paths to those in England, some features of this new approach are now becoming embedded in flood risk management policy at the level of the European Union (EU), rather than just nationally. This is most notably the case with the European Directive on the Assessment and Management of Flood Risk, which entered into force on 26 November 2007. The Floods Directive (as it is commonly known) sets out a framework for delivering improved flood risk management in all 27 EU member states. The immediate impetus behind the new Directive lies in the significant flooding in central Europe in the preceding decade, which led to pressure on the European Commission to initiate action on flooding (Samuels 2008), but its gestation also coincided with rapidly evolving thinking about the management of flooding and flood risk.

The Directive therefore covers all sources of flooding (not just rivers, but coastal floods, urban and groundwater floods). It requires planning at a basin scale and has specific requirements for international basins; and in all cases, the potential impacts of climate change on the flood conditions need to be considered. By late 2011 preliminary flood risk assessments should be in place in all European river basins, and by late 2013 there will be flood risk maps in all areas with significant risk. Flood risk management plans are to be in place by late 2015; all these are important developments.

These wide-ranging developments in flood risk management in Europe are becoming increasingly linked with broader activity in river basin management, driven by the Water Framework Directive (WFD). This came into force in late 2000 and provides a basis for the management of the ecological status of water bodies, and it includes flood management although not as a primary objective. The links between the WFD and the Floods Directive are fully recognized in the Floods Directive with the requirement to use the same boundaries and administrative structures wherever possible.

The Floods Directive seeks a common European denominator, and hence sets a minimum framework for flood risk management, which is to be interpreted in the context of each of the member states where, in many cases, concepts of flood risk management have been developing for many years. Thus in the aftermath of the severe Rhine River flooding of 1993 and 1995, the Dutch government adopted a flood policy of 'more room for rivers' with an emphasis on establishing new storage and conveyance space. In the UK the Future Flooding project (Evans *et al.* 2004) stimulated the government's 'Making Space for Water' policy (Defra 2005). In France there has been a series of initiatives to emphasize risk management rather than flood management, through an emphasis on spatial planning (Pottier 2005). There has been corresponding progressive evolution of floodplain management in the USA (Interagency Floodplain Management Review Committee 1994; Galloway 2005; Kahan 2006).

Compelling as the promise of modern integrated flood risk management certainly is, it brings with it considerable complexity. The risk-based approach involves analysing the likely impacts of flooding under a very wide range of conditions and the effect of a wide range of mitigation measures. As the systems under consideration expand in scope and timescale, so too does the number of potential uncertainties and uncertain variables. There are many potential components to a portfolio of 'hard' and 'soft' flood risk management measures, and they can be implemented in many different sequences through time, so the decision space is potentially huge. Communicating risks and building the consensus that is necessary to engage effectively with stakeholders in flood risk management requires special aptitude for communication, facilitation and mediation (Faulkner *et al.* 2007).

Characteristics of Modern Flood Risk Management

It has long been recognized that 'risk' is a central consideration in providing appropriate flood protection and latterly in flood risk management. In the UK, the Waverley Report (Waverley Committee 1954) following the devastating east coast floods of 1953 recommended that flood defence standards should reflect the land use of the protected area, noting urban areas could expect higher levels of protection than sparsely populated rural areas (Johnson 2005).

However, the practical process of flood defence design, whilst having probabilistic content, was not fundamentally risk based, proceeding somewhat as follows:

1 establishing the appropriate standard for the defence (e.g. the '100-year return period' river level), based on land use of the area protected, consistency and tradition;

2 estimating the design load, such as the water level or wave height with the specified return period;

3 designing (i.e. determining the primary physical characteristics such as crest level or revetment thickness) to withstand that load;

4 incorporating safety factors, such as a freeboard allowance, based on individual circumstances.

Meanwhile, as flood warning systems were progressively introduced and refined in the decades since the 1950s, the decision-making process was also essentially deterministic, based on comparing water level forecasts with levels that would trigger the need for and the dissemination of a warning.

Over the last two decades the limitations of such an approach in delivering efficient and sustainable flood risk management have become clear. Because ad hoc methods for decision-making have evolved in different ways in the various domains of flood risk management (flood warning, flood defence design, land use planning, urban drainage, etc.), they inhibit the integrated systems-based approach that is now promoted.

That systems approach is motivated by the recognition that there is no single universally effective response to flood risk (Proverbs 2008). Instead, portfolios of flood risk management measures – be they 'hard' structural measures such as construction of dikes, or 'soft' instruments such as land use planning and flood warning systems – are assembled in order to reduce risk in an efficient and sustainable way. The makeup of flood risk management portfolios is matched to the functioning and needs of particular localities and should be adapted as more knowledge is acquired and as systems change.

But there are institutional implications here. Implementing this approach involves the collective action of a range of different government authorities and stakeholders from outside government. This places an increasing emphasis upon effective communication and mechanisms to reach consensus. In this portfolio-based approach, risk estimates and assessments of changes in risk provide a vital common currency for comparing and choosing between alternatives that might contribute to flood risk reduction (Dawson *et al*. 2008).

The principles of flood risk assessment have become well established (CUR/TAW 1990; Vrijling 1993; USACE 1996; Goldman 1997) and are dealt with in more detail later in this volume. It is worth reviewing here how the risk-based approach addresses some of the main challenges of analysing flooding in systems (Sayers *et al*. 2002):

1 Loading is naturally variable: The loads such as rainfall and marine waves and surges on flood defence systems cannot be forecast beyond a few days into the future. For design purposes, loads have to be described in statistical terms. Extreme loads that may never have been observed in practice have to be accounted for in design and risk assessment. Extrapolating loads to these extremes is uncertain, particularly when based on limited historical data and in a changing climate.

2 Load and response combinations are important: The severity of flooding is usually a consequence of a **combination** of conditions. So, for example, overtopping or breach of a sea defence is usually a consequence of a combination of high waves and surge water levels, rather than either of these two effects in isolation. In complex river network systems the timing of rainfall and runoff at different locations in the catchment determines the severity of the flood peak. The severity of any resultant flooding will typically be governed by the number of defences breached or overtopped, as well as the vulnerability of the assets and preparedness of the people within the flood plain. Therefore, analysis of loads and system response is based on an understanding of the probability of combinations of random loading conditions and the system's responses, including the human dimension. Improved understanding of system behaviour has illustrated the importance of increasingly large combinations of variables.

3 Spatial interactions are important: River and coastal systems show a great deal of spatial interactivity. It is well recognized that construction of flood defences or urbanization of the catchment upstream may increase the water levels downstream in a severe flood event. Similarly, construction of coastal structures to trap sediment and improve the resistance of coasts to erosion and breaching in one area may deplete beaches downdrift (Dickson *et al.* 2007; Dawson 2009) and exacerbate erosion or flooding there, leading to economic damage or environmental harm. These interactions can be represented in system models, but engineering understanding of the relevant processes, particularly sedimentary processes over long timescales, is limited. Even where we have a detailed understanding of the physical processes, there may be fundamental limits to our ability to predict behaviour due to the chaotic nature of some of the relevant processes and loading.

4 Complex and uncertain responses must be accommodated: Models of catchment processes are known to be highly uncertain due to the complexity of the processes involved and the scarcity of measurements at appropriate scales (Beven 2006). The response of river, coast and man-made defences to loading is highly uncertain. The direct and indirect impacts of flooding depend upon unpredictable or perverse human behaviours for which relevant measurements are scarce (Egorova *et al.* 2008).

5 Flooding systems are dynamic over a range of timescales: Potential for long-term change in flooding systems, due to climate and socioeconomic changes, adds further uncertainty as one looks to the future. Change may impact upon the loads on the system, the response to loads or the potential impacts of flooding. It may be due to natural environmental processes, for example, long-term geomorphological processes, dynamics of ecosystems, or intentional and unintentional human interventions in the flooding system, such as floodplain development. Social and economic change will have a profound influence on the potential impacts of flooding and the way they are valued, which will be different in different countries owing to cultural factors or institutional differences.

To add further complexity, the term 'flood risk' is used today in a number of different ways. A range of meanings derived from either common language or the technical terminology of risk analysis are in use (Sayers *et al.* 2002). These different meanings often reflect the needs of particular decision-makers – there is no unique specific definition for flood risk and any attempt to develop one would inevitably satisfy only a proportion of risk managers. Indeed, this very adaptability of the concept of risk may be one of its strengths.

In all of these instances, however, risk is thought of as a combination of the chance of a particular event and the impact that the event would cause if it occurred. Risk therefore has two components – the chance (or **probability**) of an event occurring and the impact (or **consequence**) associated with that event. Intuitively it may be assumed that risks with the same numerical value have equal 'significance' but this is often not the case. In some cases the significance of a risk can be assessed by multiplying the probability by the consequences. In other cases it is important to understand the nature of the risk, distinguishing between rare, catastrophic events and more frequent less severe events. For example, risk methods adopted to support the targeting and management of flood warnings represent risk in terms of probability and consequence, but low probability/high consequence events are treated very differently to high probability/low consequence events. The former can be catastrophic leading to substantial loss of life, whereas the latter are frequent 'nuisances'. But numerical risk values are not the end of the story: other factors affecting risk and response include how society or individuals perceive that risk (a perception that is influenced by many factors including, e.g., the knowledge of recent flood events and availability and affordability of mitigation measures).

The consequences of flooding include the direct damage caused by flooding and the indirect disruption to society, infrastructure and the economy. Whilst the primary metric of the consequences is economic, the social, health and environmental effects of flooding are well recognized (Smith and Ward 1998). Thus, full descriptions of flood risk will be expressed in multi-attribute terms. Moreover, flood risk analysis problems invariably look into the future, so risk analysis involves weighing up streams of benefits and costs, which introduces problems of time-preferences. Whilst this is routinely dealt with by discounting of risks that are expressed in economic terms, the limitations, particularly for intergenerational issues, are well known (Shackle 1961; French 1988).

The benefit of a risk-based approach – and perhaps what above all distinguishes it from other approaches to design or decision-making – is that it deals with **outcomes**. Thus in the context of flooding it enables intervention options to be compared on the basis of the impact that they are expected to have on the frequency and severity of flooding in a specified area at some future date. A risk-based approach therefore enables informed choices to be made based on comparison of the expected outcomes and costs of alternative courses of action. This is distinct from, for example, a standards-based approach that focuses on the severity of the load that a particular flood defence is expected to withstand and the design of schemes to match that load.

Flood Risk Management Decisions

Flood risk management is a process of decision-making under uncertainty. It involves the purposeful choice of flood risk management plans, strategies and measures that are intended to reduce flood risk.

Hall *et al.* (2003c) define flood risk management as 'the process of data and information gathering, risk assessment, appraisal of options, and making, implementing and reviewing decisions to reduce, control, accept or redistribute risks of flooding'. Schanze (2006) defines it as 'the holistic and continuous societal analysis, assessment and reduction of flood risk'. These definitions touch upon several salient aspects of flood risk management:

- a reliance upon rational analysis of risks;
- a process that leads to acts intended to reduce flood risk;
- an acceptance that there is a variety of ways in which flood risk might be reduced;
- a recognition that the decisions in flood risk management include societal choices about the acceptability of risk and the desirability of different options;
- a sense that the process is continuous, with decisions being periodically reviewed and modified in order to achieve an acceptable level of risk in the light of changing circumstances and preferences.

Table 1.1 summarizes the range of flood risk management actions that flood risk analysis might seek to inform. It summarizes attributes of the information that is required to inform choice. So, for example, national policy analysis requires only approximate analysis of risks, though at sufficient resolution to allow the ranking of alternative national-level policies.

So, we do not need to know everything at every scale. Indeed, one of the principles of risk-based decision-making is that the amount of data collection and analysis should be proportionate to the importance of the decision (DETR *et al.* 2000). In selecting appropriate analysis methods, the aptitude of decision-makers to make appropriate use of the information provided is also a key consideration: so, for example, for flood warning decisions, timeliness is of paramount importance (Parker *et al.* 2007a, 2007b); for insurance companies, the magnitude of maximum possible losses is of central concern (Treby *et al.* 2006). The outputs of analysis therefore need to be customized to the needs and aptitudes of the different categories of decision-makers.

In Table 1.1 there is an approximate ordering of decisions on the basis of the spatial scale at which they operate. National policy decisions and prioritization of expenditure require broad scale

Table 1.1 Scope of flood risk management decisions (Hall *et al.*, 2003c)

Decision	Precision of information required	Requirement for dependable information	Spatial scope of decision	Tolerable lead-time to obtain information	Timescale over which decision will apply	Technical aptitude of decision-makers and stakeholders
National policy	Approximate	Must reflect year-on-year changes in performance	National	Months	From annual budgets to policies intended to apply over decades	Politicians advised by civil servants
Catchment and shoreline management planning	Approximate	Must be able to distinguish broad strategic options	Regional, catchment	Months to years	Sets regional policies intended to apply over decades Roughly 5-yearly review	Technical officers, but a range of non-technical stakeholders
Development control	Detailed	Consistency is expected	Local and regional development plans	Months	Decisions very difficult to reverse	Planners
Project appraisal and design	Very detailed	Costly decisions that are difficult to reverse	Local, though impacts may be wider	Months to years	Decades	Engineering designers
Maintenance	Detailed	Need to set maintenance priorities	Local; regional prioritization	Weeks	Months to years	Maintenance engineers and operatives
Operation	Very detailed	Can have a major impact on flood severity	Local	Hours	Hours	Flood defence engineers and operatives
Flood warning	Very detailed	Missed warnings can be disastrous False alarms undesirable	Regional	Hours	Hours	Flood warning specialists
Risk communication	Detailed	Inaccurate information will undermine trust	Local to national	Hours (evacuation) to years (property purchase and improvement)	Days to years	General public

analysis of flood risks and costs. This leads to a requirement for national scale risk assessment methodologies, which need to be based upon datasets that can realistically be assembled at a national scale (Hall *et al.* 2003a). Topographical, land use and occupancy data are typically available at quite high resolutions on a national basis.

The logical scale for strategic planning is at the scale of river basins and hydrographically self-contained stretches of coast (the latter from a sedimentary point of view). At this scale (Evans *et al.* 2002), there is need and opportunity to examine flood risk management options in a location-specific way and to explore spatial combinations and sequences of intervention. Decisions to be informed include land use planning, flood defence strategy planning, prioritization of maintenance and the planning of flood warnings. The datasets available at river basin scale are more manageable than at a national scale and permit the possibility of more sophisticated treatment of the statistics of boundary conditions, the process of runoff and flow, the behaviour of flood defence systems and the likely human response.

At a local scale, the primary decisions to be informed are associated with scheme appraisal and optimization, taking a broad definition of 'scheme' to include warning systems, spatial planning and perhaps temporary flood defences. This therefore requires a capacity to resolve in appropriate detail the components that are to be addressed in the design and optimization or engineering structures, or in the development and deployment of non-structural alternatives or complementary measures.

Implicit in this hierarchy of risk analysis methods is a recognition that different levels of analysis will carry different degrees of associated uncertainty. Similarly, different decisions have very different degrees of tolerance of uncertainty. Policy analysis requires evidence to provide a ranking of policy options by their efficiency or effectiveness, which can be based on approximations, whilst engineering optimization yields design variables that are to be constructed to within a given tolerance: if loss of life is threatened in that context, we need maximum precision and minimum uncertainty. We therefore now address more explicitly how uncertainty is accommodated in flood risk management decisions.

Responding to Change

It is increasingly recognized that flooding systems are subject to change on a very wide range of timescales. Whilst global climate change is most often cited as the driving force behind these processes of change (Milly *et al.* 2008), the UK Foresight Future Flooding Project (Evans *et al.* 2004) identified a host of drivers of future change. A driver of change is any phenomenon that may change the time-averaged state of the flooding system (Hall *et al.* 2003b; Evans *et al.* 2004; Thorne *et al.* 2007). Some of these drivers will be under the control of flood managers, for example construction and operation of flood defence systems, or introduction of flood warning systems to reduce the consequences of flooding (i.e. reduce the number of human receptors). Many other drivers, such as rainfall severity, or increasing values of house contents, are outside the control of flood managers and even government in general. The distinction between these two types of driver is not crisp and in terms of policy relates to the extent to which government has power to influence change and the level of government at which power is exercised. For example, decisions regarding local flood management and spatial planning are devolved to local decision-makers, whereas decisions to limit emissions of greenhouse gases are taken at national and international levels.

The range of drivers that may influence flooding systems was surveyed in the UK Foresight Future Flooding project. The drivers identified in that project as being of relevance to fluvial flooding are reproduced in Table 1.2. The Foresight study (Evans *et al.* 2004) went on to rank drivers of change in terms of their potential for increasing flood risk in the future, in the context of four different socioeconomic and climate change scenarios. Whilst the ranking was based largely upon expert judgement and a broad scale of quantified risk analysis, it did provide some indications of the

Table 1.2 Summary of drivers of change in fluvial flooding systems (adapted from Hall *et al.* 2003b)

Driver set	Drivers	SPR classification	Explanation
Catchment runoff	Precipitation	Source	Quantity, spatial distribution of rainfall and intensity. Rain/snow proportion
	Urbanization	Pathway	Changes in land surface (e.g. construction of impermeable surfaces and stormwater drainage systems)
	Rural land management	Pathway	Influences the function of surface and subsurface runoff. Changes include the proportion of conservation/recreation areas and wetlands
Fluvial processes	River morphology and sediment supply	Pathway	Changes in river morphology that influence flood storage and flood conveyance
	River vegetation and conveyance	Pathway	Changes in river vegetation extent and type, e.g. in response to climate change or due to changed maintenance or regulatory contraints
Societal changes	Public behaviour	Pathway	Behaviour of floodplain occupants before, during and after floods can significantly modify the severity of floods
	Social vulnerability	Receptor	Changes in social vulnerability to flooding, e.g. due to changes in health and fitness, equity and systems of social provision
Economic changes	Buildings and contents	Receptor	Changes in the cost of flood damage to domestic, commercial and other buldings and their contents (e.g. due to increasing vulnerability of domestic and commerical goods or increasing domestic wealth)
	Urban vulnerability	Receptor	Changes in the number and distribution of domestic, commerical and other buildings in floodplains
	Infrastructure	Receptor	Systems of communication (physical and telecommunication), energy distribution, etc. Changes in the extent to which society is dependent on these systems
	Agriculture	Receptor	Changes in the intensity and seasonality of agriculture, including removal of agricultural land from production and hence changes in vulnerability to flood damage

relative importance of different drivers of change for flood managers in the future.

The implications of change within flooding systems are profound. Milly *et al.* (2008) observe that water management decisions – their discussion was of water management in general rather than flood risk management in particular – can no longer proceed under the assumption that 'the idea that natural systems fluctuate within an unchanging envelope of variability'. The stationarity-based assumptions that have underpinned engineering design and, in our case, flood risk management are therefore no longer valid. Consequently there is a need for adaptive policies that can deliver effective risk management without relying upon untenable assumptions of an unchanging environment.

This implies a need for better models to represent these changing conditions and better observations with which to parameterize models. A recent study for the UK Environment Agency (Wheater *et al.* 2007) indicated that, to address these processes of long-term change, a new holistic modelling framework is needed, to encompass the following:

- quantitative scenario modelling of the drivers and pressures that impact upon flood risk, including global climate and socioeconomic change;
- whole catchment and shoreline modelling of flood and erosion risks under uncertain future climatic and socioeconomic conditions, and under a wide range of policy and human response options;

• integrated assessment of portfolios of response options based on economic, social and environmental criteria, including measures of vulnerability, resilience, adaptability and reversibility;
• integration of technical and socioeconomic modelling through agent-based modelling approaches;
• quantification of the various sources of uncertainty and their propagation through the modelling/decision-making process;
• a capacity for supporting a multi-level participatory stakeholder approach to decision-making.

More profoundly, the recognition of the uncertain nature of long-term change in flooding systems requires a reformulation of decision problems in order to identify options that are reasonably robust to the uncertainties surrounding future changes, where a robust option is one that performs acceptably well for a wide range of possible future conditions (Hall and Solomatine 2008).

Policy and Human Dimensions of Flood Risk Management

Uncertainty in risk assessment and the effectiveness and efficiency of policy response does not end with the natural or physical elements of the flood system. The human dimensions also embody uncertainty, and have to be analysed carefully. In that respect there has been increasing recognition over the last several decades that flood risk management is about managing human behaviour as much as managing the hydrological cycle.

Governance changes

Policy is enshrined in the institutions of governance, and the governance arrangements for flood risk management have changed many times over the last two decades in the UK (Defra 2005; Johnson 2005). This has often led to public uncertainty and confusion as to 'who is in charge'. The most recent changes have been a reduction in the influence of 'local people', who used to be represented on Regional Flood Defence Committees operating at a regional scale. The Environment Agency (EA), as the national body with flood risk management responsibilities (but only with permissive powers), is now more clearly 'in charge' but is, for some, a distant body without local accountability (House of Commons 2008). The EA, moreover, is set to obtain wider powers under legislation for England in 2010, and this may well exacerbate this sense of unease about the local flood problems of local people being misunderstood by a nationally focused and 'distant' organization. Continuing difficulties with the interaction of spatial planning and flood risk management – with continuing floodplain development in certain locations – adds to these governance issues (Penning-Rowsell 2001; Richards 2008). The fact that these issues are just as acute in the USA (Burby 2000, 2001) is no consolation to those flood 'victims' who do not know which way to turn for assistance.

Uncertainty as to response effectiveness

As we move away from flood defence and towards flood risk management – with its portfolios of measures – so the outcomes of interventions become less certain. A flood wall subject to a load it can withstand is 'safe', and can be seen to be safe, but a flood warning system may involve messages not getting through and advice that is poorly understood (Parker *et al*. 2007a, 2007b). The public's behaviour in response to flood warnings may not be what is expected by those developing the forecasts and giving the warning (Penning-Rowsell and Tapsell 2002; Parker *et al*. 2009), and a standardized approach to flood warning message design and dissemination methods – from a national body such as the Environment Agency with a national focus – may not resonate with the kind of informal arrangements that have been effective in the past (Parker and Handmer 1998). The public may be reluctant to accept measures that do not have a strong engineering focus, and therefore are seen to 'protect' them rather than just reduce the risk that they face (McCarthy 2008).

Uncertainty also surrounds the world of flood insurance in the UK. By far the majority of householders in the UK are insured against flood losses by private insurance companies. This does not mean that all losses are covered, because many of those insured are underinsured and, of course, none of the so-called 'intangible' losses from floods (Tapsell 2002) are covered at all. But it does mean that insurance is widespread. Based on the government's Household Expenditure Survey and evidence from its own members, the Association of British Insurers (ABI) estimates that the take-up of insurance in the UK is such that 93% of all homeowners have buildings insurance cover, although this falls to 85% of the poorest 10% of households purchasing their own home (where this insurance is a standard condition of a UK mortgage). Some 75% of all households have home contents insurance, although half of the poorest 10% of households do not have this cover.

But the provision of flood insurance into the future is uncertain (Arnell 2000). Previous agreements between the Association of British Insurers and the government, designed to promote flood insurance, have been renegotiated (Green *et al.* 2004; Treby *et al.* 2006). There is a distinct risk that insurance companies may withdraw from the market if government cannot continue its level of investment in flood defence projects (ABI 2005).

'Social' issues

The social effects and loss of life in floods also remain uncertain, despite considerable research effort over the last decade (Tapsell *et al.* 2002). Whilst emergency response arrangements (Penning-Rowsell and Wilson 2006) have improved massively in this time (starting with poor efforts in the UK in 1998 and developing into a much better performance through to the 2009 floods), nevertheless the social impacts of floods in traumatizing people and communities continues. Despite research into the causes of deaths in floods (Penning-Rowsell 2005; Jonkman and Penning-Rowsell 2008), loss of life in major UK floods remains a distinct likelihood. Disaster scenarios also remain a distinct possibility, especially in our large metropolitan areas (Parker and Penning-Rowsell 2005). There is a debate to be had about what flood risk management measures are the fairest (Johnson 2007b), but the available research shows that the poor and disadvantaged suffer most in events such as floods (Walker 2003), owing to their lack of savings, insurance and the wherewithal or knowledge as to how to protect themselves.

But modern flood risk management is people-focused. Considerable emphasis is now placed on stakeholder attitudes and aspirations, with government and state agencies alike seeking public engagement in the decisions that affect them, decisions that require behavioural change for effective implementation (not something that is generally needed when tackling floods with concrete walls but that is needed when seeking an efficient public response to a flood warning).

However, it remains true that public attitudes are fickle and risk remains very poorly understood (Faulkner *et al.* 2007). Immediately after a flood the demands are for 'action', and for blame to be accepted by those 'in charge'. Five years later the public is antagonistic when those very same people 'in charge' produce designs for a flood defence scheme, or promote tighter spatial planning rules, which might restrict regenerative developments at a time when such economic revival is a local imperative. Memories are short, denial is a common theme, and the public has many other issues about which to worry. Conflict is almost inevitable, with all the further uncertainty that this is likely to bring.

A Blueprint for Modern Flood Risk Management

Understanding of the process of flood risk management continues to evolve. The contributions in this volume represent various dimensions of the state of the art. Yet it would misunderstand the nature of flood risk management if it were taken to be a fragmented set of techniques – far from it, flood risk management entails a systems perspective, which is itself embedded within the broader perspectives of sustainable development. Here we highlight a number of pertinent aspects not only of where flood risk management is now, but where it may be going in the future.

• **Risk based:** Flood risk management is by definition risk based! The reason is that this provides a rational basis for comparing management options. However, as we have seen already, evaluating the likelihood and consequences of flooding now and in the future is fraught with difficulties. We can in future expect more scientific estimation of the probabilities of relevant flooding processes, on a range of different timescales. Methods for better evaluation of the consequences of flooding and the side-effects of flood risk management, on a range of timescales, are urgently needed in practice and, we expect, will be taken up enthusiastically as soon as they are well tested by the research community.

• **Systems based:** The nature and interactions of multiple sources of flooding are beginning to be understood. Surface water flooding in urban areas may not be as devastating as a coastal dike breach, but it occurs much more frequently and can be disruptive to economic activity and society and can cause loss of life. Thus all of these flooding mechanisms need to be managed in an integrated way. Arbitrary subdivision of the flooding system, for example due to geographical boundaries or administrative divisions, is to be avoided.

• **Portfolio based:** Integrated management involves consideration of the widest possible set of management actions that may have some impact on flood risk. This includes measures to reduce the probability of flooding and measures to reduce flood impact (exposure and vulnerability) and development of integrated strategies. Management strategies are developed following consideration of both effectiveness, in terms of risk reduction, and cost. They will involve coordinating the activities of more than one organization and multiple stakeholders.

• **Multi-level:** Flood risk management cascades from high-level policy decisions, based on outline analysis, to detailed designs and projects or measures, which require more detailed analysis. High-level policy and plans provide the framework and common understanding within which more detailed actions are implemented.

• **Evidence based:** Flood risk management is often dealing with situations and scenarios that have never occurred in practice. It relies therefore on statistical and physically based predictive modelling. Advances in this modelling capacity have underpinned the introduction of the flood risk management paradigm, as has the broader paraphernalia of geographic information systems (GIS) and decision support systems. These powerful tools need to be soundly based upon empirical evidence. The path of analysis from empirical evidence to risk-based recommendations should be visible and open to scrutiny.

• **Robust:** We have discussed above the impact that uncertainty can have on flood risk management decisions. Uncertainty analysis should therefore not only be central to the process of conducting flood risk analysis but should also underpin the formulation of flood risk management decisions and the evaluation of responses.

• **Adaptive:** Flood risk management has to explicitly recognize change in flooding systems on a range of timescales and due to a variety of processes. This will involve recasting many statistical analyses and a renewed emphasis upon physically based models (and necessary empirical observations) that can represent processes of change. It implies a commitment to careful monitoring of processes of change, including socioeconomic change. More fundamentally, it will place more emphasis upon the capacity of decision-makers to deal with irreducible uncertainty and of designers to innovate solutions that are flexible and adaptable in future.

• **People based and democratic:** A host of different stakeholders have an interest and role in the process of flood risk management. Successful risk reduction relies, to some extent, upon the engagement of stakeholders in raising awareness of flood risk, emergency management and recovery. The impacts of floods include serious social and human harm, and local people may be valuable providers of local knowledge to help with implementing effective risk reduction measures. More broadly, people at risk from flooding have a legitimate interest in the decisions that are being taken on their behalf. Thus effective flood risk management involves engagement of stakeholders throughout the decision-making processes and relies upon proper processes and resources being in place to manage that stakeholder engagement.

• **Integrated within sustainable development:** We have mentioned above the relationship between the Water Framework Directive and the Floods Directive. Flooding is one of the functions of river basins and coastal systems, and flood risk management is one dimension of integrated water resource management. Flood risk management also forms part of the broader process of preparing for and limiting the impacts of natural and human-induced hazards.

We can see that some but not all of the dimensions of flood risk management that are mentioned above are fully developed in government or institutional thinking. They will be interpreted in different ways in different national contexts and in relation to the nature of the different flooding systems. However, remarkable progress in cultivating these concepts has taken place over the past decade and in many instances this progress has been transferred into decision-making practice. This volume seeks to ride this synergistic wave of innovation in research, policy and practice, and in the following sections various dimensions of flood risk management are expanded upon and illustrated with case studies, in order to populate the broad framework for flood risk management that has been set out in this introductory chapter.

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Part 2
Land Use and Flooding

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2 Strategic Overview of Land Use Management in the Context of Catchment Flood Risk Management Planning

ENDA O'CONNELL, JOHN EWEN AND GREG O'DONNELL

Strategic Approach to Flood Risk Management Under Changing and Uncertain Conditions

It is widely recognized that, to cope with the impacts on flooding of climate variability and change, holistic approaches to managing flood risk are needed, as are new integrated research frameworks that can support these new approaches. The Office of Science and Technology (OST) Future Flooding project (Evans *et al*. 2004a, 2004b, 2008) developed the thinking for a holistic approach to managing flood risk, which was taken on board in formulating the government's strategy for managing flood and coastal erosion risk in England – 'Making Space for Water (MSW)' (Defra 2004). This MSW approach is risk-driven and requires that adaptability to climate change is an integral part of all flood and coastal erosion management decisions. A whole-catchment approach is being adopted that is consistent with, and contributes to, the implementation of the Water Framework Directive (2000/60/EC). The MSW strategy requires the consideration of a broad portfolio of response options for managing risks including changes to land use planning in flood-prone areas, urban drainage management, rural land management and coastal management. Stakeholders are engaged at all levels of risk management, with the aim of achieving a better balance between the three pillars of sustainable development (economic, social and environmental) in all risk management activities (Defra 2004).

To support this integrated approach to flood risk management, it is evident that a corresponding integrated approach to catchment planning is needed that can support the implementation of the MSW strategy over the next 20 years and beyond. Heretofore, catchment modelling has been technical and compartmentalized, has assumed that the past climate is representative of the future, and has not quantified the different sources of uncertainty in the modelling and decision-making process; nor has it considered the full socioeconomic context. The integrated modelling framework must therefore encompass the following (Wheater *et al*. 2007):

- quantitative scenario modelling of the drivers and pressures that impact upon flood risk, including global climate and socioeconomic change;
- whole catchment and shoreline modelling of flood and erosion risks under uncertain future climatic and socioeconomic conditions, and under a wide range of response options;

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• integrated assessment of portfolios of response options based on economic, social and environmental criteria, including measures of vulnerability, resilience, adaptability and reversibility;
• integration of technical and socioeconomic modelling through agent-based modelling approaches;
• quantification of the various sources of uncertainty and their propagation through the modelling/decision-making process;
• the capability to support a multi-level participatory stakeholder approach to decision-making.

All of the above can be represented within the Driver-Pressure-State-Impact-Response (DPSIR) logical framework, which is used widely in integrated environmental and socioeconomic studies of environmental change. The DPSIR framework, and variants thereof, has been applied in a number of recent studies relating to flooding and coastal management. For example, Turner *et al.* (1998) used a DPSIR framework to analyse environmental and socioeconomic changes on the UK coast, and the framework was also used within the OST Future Flooding project (Evans *et al.* 2004a, 2004b, 2008).

As part of a review of the impacts of rural land use and management on flood generation (Project FD2114), O'Connell *et al.* (2004, 2005) employed the DPSIR framework to describe the broad anthropogenic context for flood generation on rural land (Fig. 2.1). This allowed the historic dimension of land use and management over time to be considered and how changes in management practices over time have given rise to concerns about flood generation. The review found that there is considerable evidence that agricultural commodity markets and agricultural policies, currently contained within the EU Common Agricultural Policy, are key **drivers** that critically influence land use management. These in turn lead to **pressures** on land and the water environment generated by intensive agriculture, associated, for

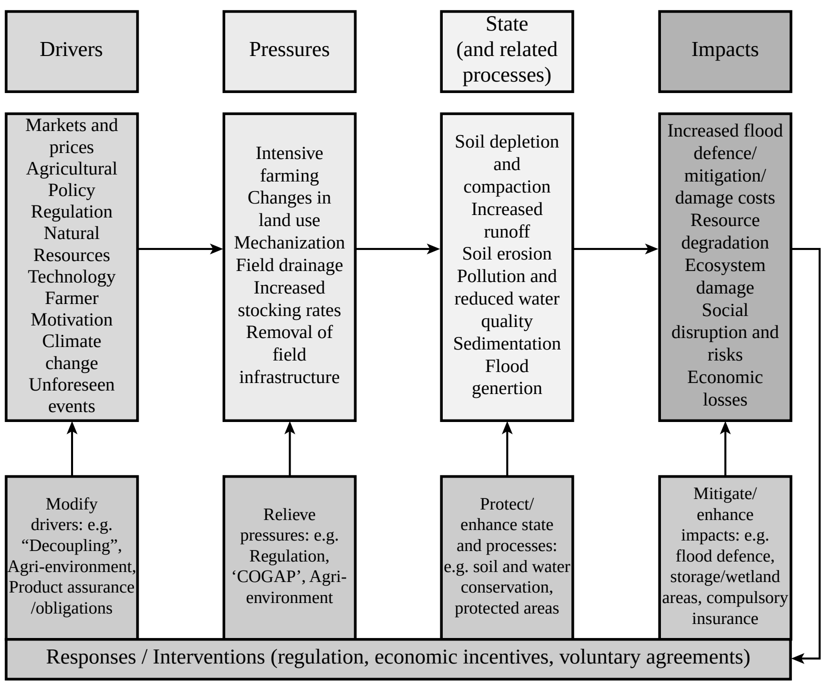

Fig. 2.1 Driver-Pressure-State-Impact-Response (DPSIR) framework applied to flood generation from rural land (O'Connell *et al.* 2004, 2005). COGAP, code of good agricultural practice.

example, with changes in land use type such as the switch from grassland to arable, changes in farming practices such as intensive mechanization within a given land use type, or changes in field infrastructure such as the installation of field drains or the removal of hedges. In turn, these pressures can change the **state** of rural catchments, reducing the integrity and resilience of environmental characteristics and processes with potential to increase runoff, soil erosion and pollution. If unchecked, this can result in negative **impacts** on people and the environment and the loss of welfare that this implies. A particular feature of runoff (and water-related soil erosion and pollution from rural land) is that impacts, when they do arise, are mainly 'external' to the site of origin and are borne by third parties usually without compensation. In this respect, land managers may be unaware of, or may have little personal interest in alleviating, the potential impacts of runoff, unless they are instructed otherwise. Concern about impacts justifies **responses** in the form of interventions that variously address high-level drivers, land management pressures, protect the state of the environment and mitigate impacts. Responses, which may involve regulation, economic incentives, or voluntary measures, are more likely to be effective, efficient and enduring where they modify drivers and pressures, rather than mitigate impacts (O'Connell *et al.* 2005).

This chapter uses the DPSIR framework as a starting point to demonstrate how land use management fits within a broad strategic research framework for flood risk management at the catchment scale. Catchment scale modelling and prediction is of central importance to assessing impacts within DPSIR, and is also central to assessing the effectiveness of mitigation response measures. The current status of the capacity to model impacts will therefore be a central feature of this chapter. First, the historical context for land use management changes is set out, and the evidence for impacts at local and catchment scales is summarized. A strategic modelling framework for flood risk management based on DPSIR is then mapped out, which includes an integrated programme of multiscale experimentation and modelling being undertaken in the Flood Risk Management Research Consortium (FRMRC) and other related research programmes. The problem of modelling and predicting impacts at the catchment scale is reviewed, and the major challenges associated with filling key gaps in knowledge are discussed. New modelling concepts such as information tracking are introduced, and their application to vulnerability mapping and Source-Pathway-Receptor modelling for policy and decision support in catchment flood risk management planning is demonstrated.

Historical Context: Runoff Generation and Routing in Changing Landscapes and the Evidence for Impacts

Changes in land use and management

Since the Second World War, the UK landscape has undergone major changes as a result of the drive for self-sufficiency in food production, and the effects of the Common Agricultural Policy:

- loss of hedgerows, and larger fields;
- cultivation practices causing soil compaction to a greater depth;
- land drains connecting the hilltop to the channel;
- cracks and mole drains feeding overland flow to drains and ditches;
- unchecked wash-off from bare soil;
- plough lines, ditches and tyre tracks concentrating overland flow;
- tramlines and farm tracks that quickly convey runoff to watercourses;
- channelized rivers with no riparian buffer zones.

In this landscape there are several interacting factors that will have induced changes in the generation of runoff and its delivery to the channel network, such as the extent of soil compaction, the efficiency of land drains, and the connectivity of flow paths. A key factor is the impact that soil structure degradation (due to compaction) can have on runoff generation. By influencing the soil structural conditions that determine the inherent storage capacity within the upper soil layers, and their saturated hydraulic conductivity, land management can significantly affect the local generation of surface and subsurface runoff. Management practices that cause soil compaction at the surface reduce the infiltration capacity of the soil and can lead to infiltration-excess runoff. Similarly, practices that leave weakly structured soils with little or no vegetative cover can also lead to infiltration-excess runoff, as a result of the rapid formation of a surface crust with very low moisture storage capacity and hydraulic conductivity. Practices that cause compaction at the base of a plough layer can also lead to saturation-excess surface runoff, and to subsurface runoff by rapid lateral throughflow in the upper soil layers. Apart from the soil degradation factors, several other factors associated with land use and management can potentially influence runoff generation. For example, the maintenance of land drains has declined since the 1980s when subsidies ceased, and many of these may have become blocked and do not function effectively.

The landscape within a catchment is a complex mosaic of elements, all with different responses and overlain by a range of land management practices, so there is the key issue of how the responses of these elements combine to generate the overall catchment response. As runoff is routed from the local scale to the catchment scale, the shape of the flood hydrograph will reflect increasingly the properties of the channel network, such as its geometry, the slopes and roughnesses of individual stretches, and attenuation induced by floodplain storage effects when out-of-bank flooding occurs. However, the magnitude of the flood peak will also reflect the volume and timing of runoff from landscape elements delivered into the channel network, and the extent to which the timings of the peaks of tributary hydrographs are in phase or out of phase with the main channel hydrograph or with each other. This will all vary as a function of the magnitude of the flood, as travel times are a function of water depth, and will depend on the spatial distribution of rainfall over the catchment.

When considering impact, therefore, the main questions are:

1 At the local scale, how does a given change in land use or management affect local-scale runoff generation?
2 How does a local-scale effect propagate downstream, and how do many different local-scale effects combine to affect the flood hydrograph at larger catchment scales?
3 How can adverse effects be mitigated using economically and environmentally acceptable measures?

Evidence for impacts and mitigation

The following is a brief summary of current knowledge about local-scale impacts at the farm plot/hillslope and catchment-scale impacts. The scope of the summary is confined primarily to UK studies, supplemented by overseas studies in temperate environments with similar land use/management practices. The findings from the overseas studies are generally in close agreement with those from the UK. Further details can be found in O'Connell *et al*. (2005, 2007).

Local-scale impacts

Local surface runoff can increase as a result of a number of modern farm management practices such as:
• increased stocking densities on grassland (UK studies: Heathwaite *et al*. 1989, 1990; USA: Rauzi and Smith 1973);
• the prevalence of autumn-sown cereals (Belgium: Bielders *et al*. 2003; UK: Palmer 2003b; Denmark: Sibbesen *et al*. 1994);
• the increase of maize crops (UK: Clements and Donaldson 2002; Netherlands: Kwaad and Mulligen 1991);
• the production of fine seedbeds (UK: Edwards *et al*. 1994; Speirs and Frost 1985);
• trafficking on wet soils (UK: Davies *et al*. 1973; France: Papy and Douyer 1991; USA: Young and Voorhees 1982).

There does not appear to be a strong link with soil type, but sandy, silty and slowly permeable seasonally wet soils are more susceptible than others. Reduced infiltration and increased surface runoff associated with modern practices are widespread (Souchere *et al.* 1998; Holman *et al.* 2001; Hollis *et al.* 2003; Palmer 2003a, 2003b).

Field-drainage and associated subsoil treatments can increase or decrease peak drain flows and the time to peak flow by as much as two to three times either way; the behaviour appears to depend on the soil type and wetness regime (Leeds-Harrison *et al.* 1982; Armstrong and Harris 1996; Robinson and Rycroft 1999).

Enhanced surface runoff generation as a result of some of the above modern farming practices can generate local-scale flooding. For example, long-term studies in small catchments in the South Downs of southeast England show that there is a significant relationship between the presence of autumn-sown cereal fields and local 'muddy floods' in autumn (Boardman *et al.* 2003). This relationship has also been observed in France (Papy and Douyer 1991; Souchere *et al.* 1998) and Belgium (Bielders *et al.* 2003). The frequency of these floods can be reduced using appropriate arable land management practices (Evans and Boardman 2003). Muddy floods, and the erosion and subsequent deposition of substantial amounts of eroded soil, generate substantial economic damages each year, most of which occur off-farm (Evans 1996).

There is, in contrast, very little direct evidence of how such changes affect the flow in surface water networks, and evidence that is available is for small catchments ($<10\,km^2$), and mainly relates to the impacts of forests, which are generally considered to reduce flood peaks, except for the effects of drainage and forest roads (McCulloch and Robinson 1993). However, peak flows can increase in the period after forest planting, mainly as a result of plough drainage and ditching (Robinson 1986; Robinson *et al.* 1998). In a review of results from 28 monitoring sites located throughout Europe, Robinson *et al.* (2003) concluded that forests probably have a relatively small role to play in managing regional or large-scale flood risk, and significant local-scale impacts are likely only for the particular case of managed plantations on poorly drained soils.

The effects on runoff of subsurface drainage are soil dependent and impacts on downstream flooding are difficult to interpret from field data. River channel improvements can have a much greater effect on peak flows than field drainage (Robinson 1990).

Catchment-scale impacts

National analyses of flooding trends have not shown significant impacts of either climate or land use change, largely because of the overriding influence of year-to-year climatic variations, which make trends associated with climate and land use difficult to identify (Robson *et al.* 1998; Institute of Hydrology 1999). The UK Flood Estimation Handbook (Institute of Hydrology 1999) is based on two methods of flood estimation, the Statistical Approach and the Rainfall-Runoff Approach. Regression relationships linking flood statistics (e.g. the median annual flood) or rainfall-runoff method parameters (e.g. the time to peak of the unit hydrograph) with catchment characteristics did not reveal any significant relationship with land cover. It should be noted, however, that the records used in the analysis were mainly from catchments not experiencing major land use change (see UK Flood Estimation Handbook, Vol. 3, p. 234), and that land cover data cannot alone reflect land use management practices.

River channels in the UK have also undergone substantial modifications over the past 70 years as a result of land drainage schemes and flood protection works for urban and rural floodplain areas (Newson and Robinson 1983; Robinson 1990; Robinson and Rycroft 1999; Sears *et al.* 2000). Channels have been subject to a number of different modifications, depending on the circumstances, for example straightening, resectioning, embanking, culverting and the construction of weirs and sluices. More recently, there has been a move towards the restoration of channels and floodplains to their natural states and functions, as part of biodiversity and natural flood mitigation schemes. It is clear that such modifications will have changed the natural routing processes in many UK catchments, so would have to be taken into account (a formidable challenge) when assessing evidence that changes to local runoff generation processes have affected flooding at the catchment scale.

Historical rainfall runoff datasets have been analysed to look for impacts of land use and management change on flood generation (Beven *et al*. 2008), for a set of six predominantly rural catchments (75–1134 km^2) which (i) were considered to be candidate catchments where land use management impacts might have taken place, and (ii) had reasonably long rainfall and runoff records available. No clear evidence for significant impacts was found, but this does not necessarily mean that impacts do not exist, only that they were not detectable for the catchments analysed, given both the natural variability of the climate and the inadequacies of the available data in characterizing the hydrological response.

Mitigation of impacts

Interventions can mitigate or avoid the impacts of land use management on local flooding. The majority of these interventions are aimed at source control of on-farm runoff through the use of good land use management practices. For example, for maize cropping, particularly in free-draining loamy, silty and sandy soils, ploughing in the autumn and spring can reduce field plot runoff significantly (Kwaad and Mulligen 1991; Martyn *et al*. 2000; Clements and Donaldson 2002). The success of other management techniques such as direct drilling, cover crops and soil mulches appears to be much more uncertain and dependent on soil type (Charman 1985; Schafer 1986; Auerswald 1998; Melville and Morgan 2001; Clements and Donaldson 2002).

Desirable management practices for mitigating field-scale runoff generation are depicted in Figure 2.2. Most require careful targeting with

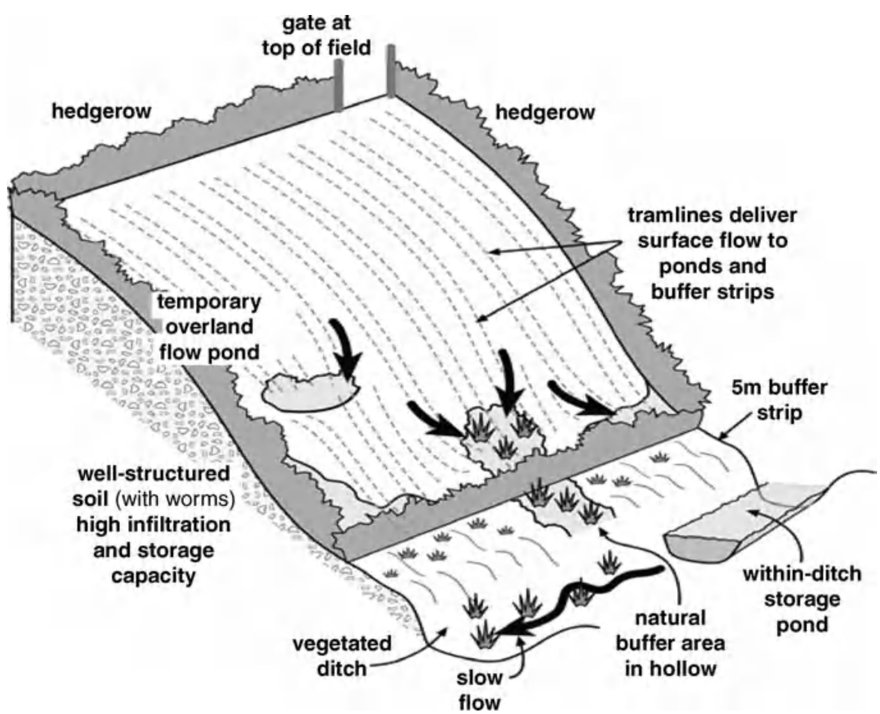

Fig. 2.2 Potential for integrated runoff control to reduce flood risk, pollution and erosion.

respect to specific topographic, soil, cropping and climatic conditions. Moreover, such measures can also control nutrient pollution and sediment transport, thus generating multiple benefits for the water environment.

Local-scale mitigation measures (e.g. at the farm scale) can be viewed as 'prevention at source', but, since their effect will essentially be to delay or attenuate the delivery of runoff (e.g. by changing the partitioning of surface and subsurface runoff through increased infiltration), the overall effect on the catchment flood hydrograph will depend on how these changes affect the hydrological functioning of the catchment as a whole, given that they will interact with other ongoing changes (e.g. to river and floodplain management).

Strategic Research Framework

In a recent review of future research requirements for flood risk management (Wheater *et al.* 2007), broad-scale modelling requirements to support decision-making in planning whole catchment and whole shoreline flood risk management were identified. The various elements of the proposed broad-scale modelling (BSM) programme were then mapped into a DPSIR framework (Fig. 2.3). Figure 2.3 illustrates that a comprehensive, integrated modelling approach must encompass not just the central impact modelling and prediction that is typically formulated to support a Source-Pathway-Receptor approach to the management of flood risk, but also consideration of the drivers and pressures that may give rise to future land use management changes. Moreover, mitigation interventions must be assessed using a broad, integrated assessment approach that considers economic, social and environmental aspects of sustainability as outlined in 'Making Space for Water'.

Figure 2.4 is a schematic of the various elements of an integrated research programme assembled to address the following priorities identified by O'Connell *et al.* (2005); these relate to understanding, modelling and predicting states, impacts and land use management response impacts within the DPSIR-BSM modelling framework in Figure 2.3:

- Detection of land use management impacts in catchment flood response data [followed up in Project FD2120 (Beven *et al.* 2008)].
- Multiscale experiments across a range of catchment scales that would cover a range of land use management interventions and provide high-quality data for model validation; these are needed to answer Question 2 above: how does a local-scale effect propagate downstream, and how do many different local-scale effects combine to affect the flood hydrograph at larger catchment scales?
- New distributed modelling approaches that would allow impacts to be tracked from local to catchment scales.
- Field trials of mitigation measures (responses) to determine their performance.
- Tools to support decision-making such as SPR modelling and vulnerability mapping.

There are four ongoing field experiments (SCaMP, CHASM, Pontbren and Belford), which are supplying high-quality data, while new modelling developments are taking place within both the Natural Environment Research Council 'Flood Risk from Extreme Events' (NERC-FREE) and FRMRC programmes that will ultimately feed through into the next generation of catchment-scale flood risk management planning (e.g. in Modelling and Decision Support Framework, MDSF). All of these are summarized below.

Multiscale Experimentation in Support of Modelling and Prediction

The CHASM multiscale experimentation programme

The importance of understanding scale effects, particularly the factors controlling the variability in response both within and between catchments, was recognized in 1998 when a consortium of universities and research institutes was formed to develop the Catchment Hydrology and Sustainable Management (CHASM) programme of integrated multiscale experimentation, modelling and prediction. In CHASM, the instrumentation of

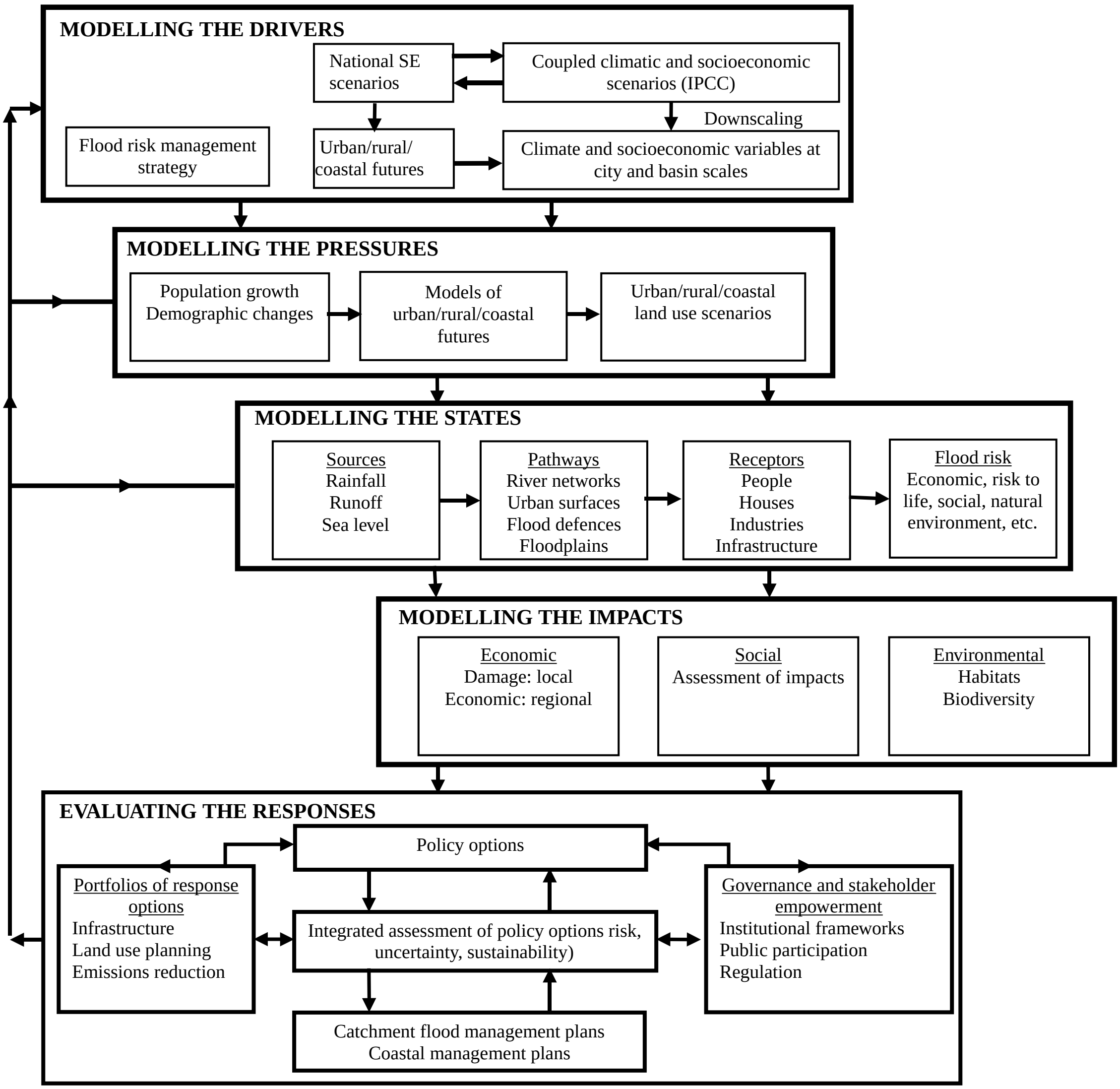

Fig. 2.3 Driver-Pressure-State-Impact-Response (DPSIR)-broad-scale modelling (BSM) Decision Support Framework.

mesoscale catchments (~100 km^2) was given a high priority (historically, this is a neglected scale in hydrological field research), and hydrological response has been monitored across a range of increasing scales, from the hillslope to the mesoscale. Funding of £2 million was obtained from the Natural Environment Research Council under the JIF (Joint Infrastructure Fund) initiative to instrument four mesoscale catchments in England, Northern Ireland, Scotland and Wales: the Upper Eden (337 + 90 km^2), Oona (92 km^2), Feshie (200 km^2) and Upper Severn (182 km^2), respectively. The catchments were selected to take advantage of existing instrumentation associated with previous small-scale catchment experiments, including the Plynlimon and Coalburn catchments,

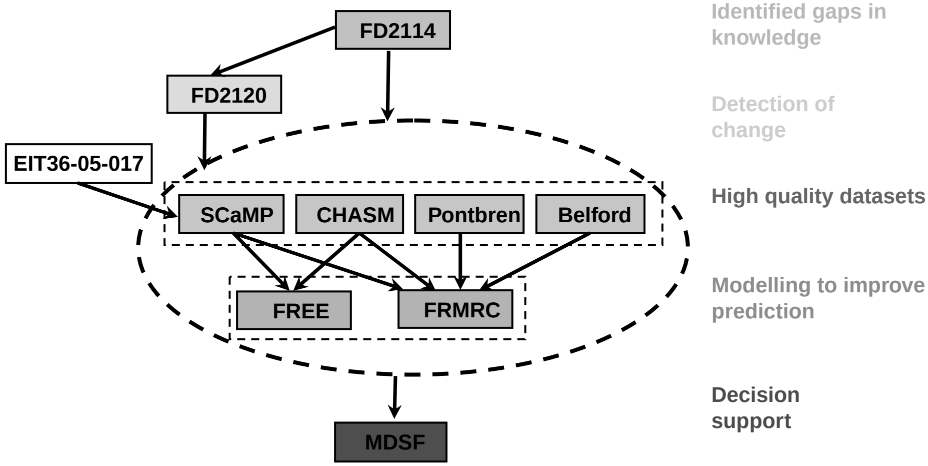

Fig. 2.4 Integrated land use management research programme. CHASM, Catchment Hydrology and Sustainable Management; FREE, Flood Risk from Extreme Events; FRMRC, Flood Risk Management Research Consortium; MDSF, Modelling and Decision Support Framework; SCaMP, Sustainable Catchment Management Plan.

and that deployed by the UK Environment Agency.

Instrumenting mesoscale catchments is difficult and expensive, so a custom-designed approach was developed, in which mobile and permanent instrumentation were used to optimum effect. In the case of the Eden catchment, the investigators have been particularly fortunate in capturing multiscale data for some major floods that occurred in 2004 and 2005 (Mayes *et al*. 2006); the 2005 flood inundated the city of Carlisle. Figure 2.5 illustrates how the hydrograph for the 2004 flood changes with increasing scale.

Although land use management changes have not been a specific focus of the CHASM programme, the resulting multiscale data for the 2004 and 2005 floods can be used for testing and validating new modelling approaches across a range of scales. The CHASM experimental design approach has also served as a blueprint for other multiscale experiments, such as in the Hodder, discussed below.

The Pontbren multiscale experiment

The Pontbren experiments are described in detail in Chapter 3. Pontbren, situated in the headwaters of the River Severn in Wales, is a farmers' cooperative concerned with sustainable upland agriculture, involving 10 hill farms and over 1000ha of agriculturally improved pasture and woodland.

The farmers' perception is that changes to land management, and in particular changes to grazing densities and animal weights, have changed runoff response. Although land use has changed relatively little since the 19th century, between the 1970s and 1990s dramatic changes in farming intensity took place; sheep numbers increased by a factor of six, and animal weights doubled.

The current scales of research range from experimental plots to an 18-km^2 catchment, including three first-order streams. The experiments focus on soil properties and runoff processes, based on plot and hillslope scale measurements nested within instrumented first- and second-order catchments (Marshall *et al*. 2008).

The SCaMP/Hodder multiscale experiment

Under the United Utilities Sustainable Catchment Management Plan (UU SCaMP) being implemented in the Upper Hodder catchment, northwest England, extensive changes are being made to land use/management – see the scoping study carried out by Newcastle University for

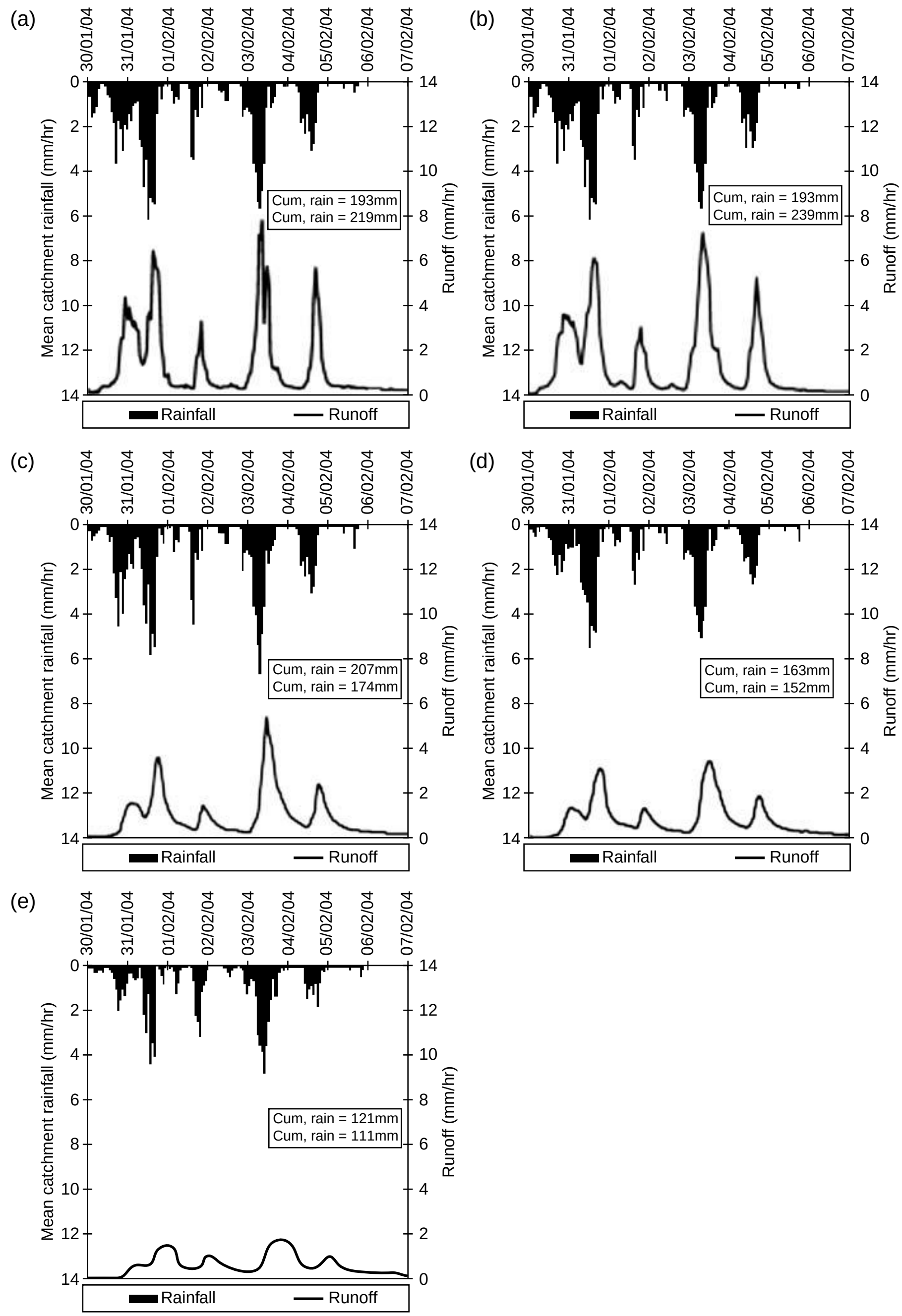

Fig. 2.5 Eden 2005 flood hydrograph at increasing scales. (a) Gais Gill (1.1 km^2); (b) Artlegarth Beck (2.7 km^2); (c) Scandal Beck (37 km^2); (d) Eden: Great Musgrave (222 km^2); (e) Eden: Temple Sowerby (616 km^2).

English Nature Project E1T36-05-017 (Ewen *et al.* 2006). The changes include woodland planting, blocking of moorland grips, changes in stocking density, changes in heather burning policy, and the creation of scrapes for wading birds, all with the aim of preventing further deterioration of raw water quality (especially water colour production in water abstracted for water supply) and improving the ecological condition of Sites of Special Scientific Interest.

The Environment Agency have funded a project (SC060092) to enable the effects of the SCaMP land use management changes on downstream flooding to be monitored and assessed. The specific objectives are to: (i) create a database that defines and stores the required data, which can be used as a general template for the data requirements in future field/modelling programmes on the effects of land use/management on local and downstream flooding; (ii) collect the necessary data for the SCaMP site and its downstream catchments, as a general resource; and (iii) run a preliminary analysis of the effect of SCaMP on local and downstream flooding.

The multiscale monitoring network implemented in the Hodder is shown in Figure 2.6.

The Belford flood mitigation experiment

The Belford study (http://www.ncl.ac.uk/iq/download/BelfordBHSpaper.pdf) has been established to provide a full operational assessment of the extent to which flooding at local scale can be mitigated to provide flood protection for a small community. The town of Belford, Northumberland, has been subject to flooding for several years,

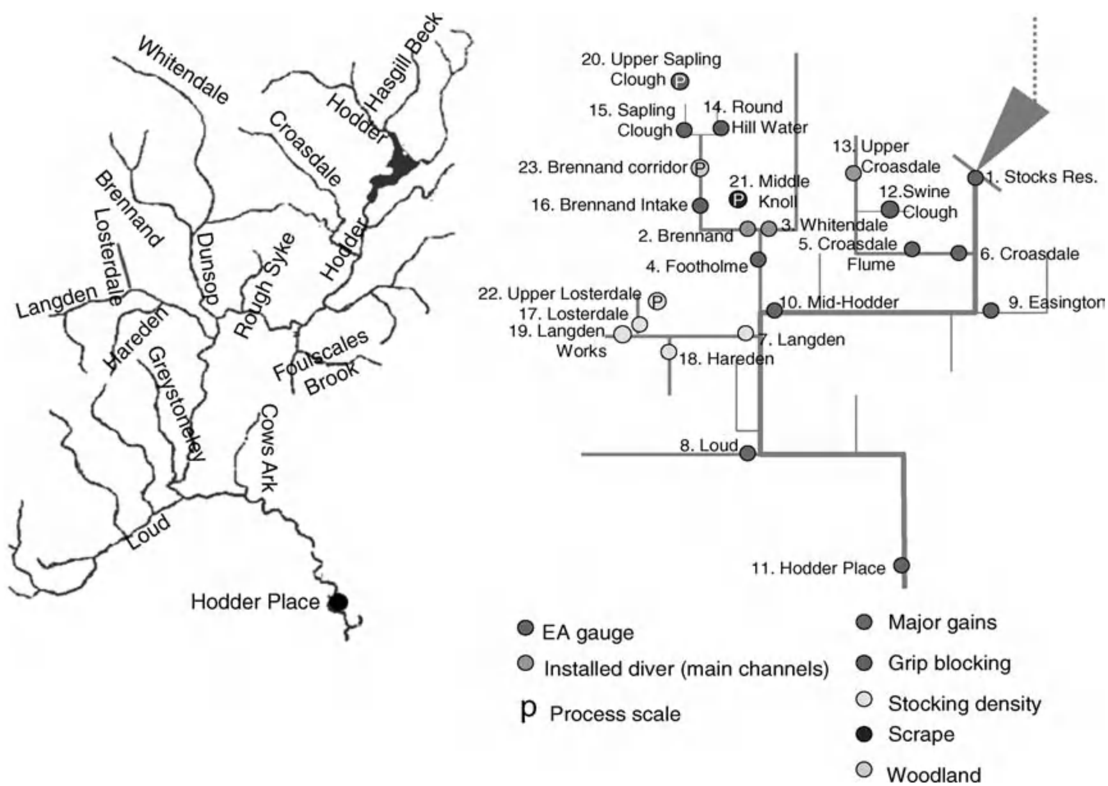

Fig. 2.6 Multiscale nested experiment in the Hodder. EA, Environment Agency. (See the colour version of this figure in Colour Plate section.)

but does not qualify for traditional EA flood protection investment based on cost-benefit analysis. Funded by the local Environment Agency Flood Levy, a number of small flood storage ponds and associated diversions have been installed that have been proven to attenuate flooding, notably for the September 2008 flood, which inundated the nearby town of Morpeth. The Belford study is yielding an abundance of good-quality data to show how runoff propagates through small rural areas and how flood waves can be attenuated by small flood storage ponds (Fig. 2.7). Further features are being installed including a large temporary storage pond, which is large enough to warrant a sluice gate (which will be operated by the farmer); a woody debris zone in a forest location, which is using locally felled trees to create beaver dams and a rough floodplain inundation area; a zone of willow barriers on the suitable small floodplain zone; and a diversion structure from the main road onto the farmers' fields and into another large pond and more small ponds in fields (10–20 in total).

Based on the data that are being collected, it is intended that a generalized model of a system of such mitigation features will be developed that would allow the flood attenuation potential of such interventions to be assessed at the catchment scale, thus widening the portfolio of land use management interventions that could be considered for catchment planning.

Fig. 2.7 On-farm runoff storage at Belford.

Modelling, Predicting Impacts and Vulnerability Mapping

Modelling issues

In an ideal world, the prediction of the likely impact of future changes in land use/management would use simple models that encapsulate knowledge derived directly from comprehensive data on observed impacts. Although new data are being collected through the experiments described above, they are site/catchment specific. To obtain predictions of flood impacts for other catchments, we must rely heavily on catchment rainfall-runoff modelling, concentrating our efforts on how hydrological processes and change effects can be represented in these models (O'Connell *et al.* 2007).

The following method is widely used to predict the impact of changes in land use/management (e.g. Bormann *et al.* 1999; De Roo *et al.* 2001; Fohrer *et al.* 2001; Niehoff *et al.* 2002; Liu *et al.* 2005; Brath *et al.* 2006):

1 Select the catchment rainfall-runoff model that has the most appropriate representation of the hydrological processes affected by the change.
2 Calibrate the model and run simulations of the catchment in its state prior to change.
3 Alter the model's parameters to reflect the change (this is where field data on change effects can be used).
4 Run simulations using the altered parameters.
5 Estimate the impact, as the difference between the simulated responses in steps 4 and 2.

On reviewing this method, and the wider problem of predicting impacts, O'Connell *et al.* (2007) concluded that it raises some questions:

1 What is the most appropriate type of model for the prediction of change?
2 Which hydrological processes need to be incorporated into a model, and in how much detail?
3 Which model parameters need to be altered to reflect a change (and how can their values be specified a priori)?
4 How can the uncertainty in the results be quantified?
5 How can a model be validated for predicting impacts?

To get right to the heart of the problem of predicting change effects, Question 3 can be broadened out into a question about how detailed knowledge and understanding gained at a small scale (e.g. in field plots where land use/management or rainfall is manipulated) finally ends up affecting predictions made for impacts at large scales (e.g. flooding of a floodplain, downstream). The reason why it broadens out in this way is that, as shown in the previous section, we are far more likely to gain information about change effects at small scales than at large scales. Perhaps if progress can be made with this broader question, then the other questions will be easier to answer.

Part of the solution must lie in creating a direct link between small-scale parameters and large-scale impacts, and making sure that nothing is done in the modelling process that corrupts or breaks this link. There are a few ways that the link can be made, including multiscale modelling where models at different scales are combined (e.g. Bronstert *et al.* 2007), and using direct transfers via explicit functions that define how large-scale parameters are related to small-scale parameters (e.g. Hundecha and Bárdossy 2004). Another approach is metamodelling (Ewen 1997; Ewen *et al.* 1999; Kilsby *et al.* 1999; Audsley *et al.* 2008; Jackson *et al.* 2008), as described in Chapter 3. The main steps in a metamodelling approach are:

1 Use models at the small scale with parameters that are, as far as practical, based directly on measurements made at the small scale (e.g. physically based runoff generation models).

2 Build a simple model that reproduces the responses produced by the small-scale model. That is, build an emulation model (this is called the metamodel).

3 Build the necessary large-scale model:

- **a** break the catchment into many areas (e.g. by grid or subcatchment);
- **b** apply the emulation model to each area;
- **c** add a routing scheme to link the areas; and
- **d** create a classification/regionalization method, so that the effort spent on emulation is reduced from finding parameters for each of the areas to finding parameters for each of the classes (assuming there are far fewer classes than areas).

As one might expect, this approach is fraught with difficulties, not least that: (i) the small-scale model must encapsulate information derived from small-scale measurements – but small-scale hydrology is never simple, particularly when natural hydrological functioning has been altered through intensive agriculture; and (ii) by its very nature, the classification/regionalization method must use some type of large-scale (approximate) information, such as maps of land cover. These difficulties, though, are not specific to the problem of predicting change impacts, but are present in some form whenever a bottom-up approach is used in catchment rainfall-runoff modelling.

Information tracking and vulnerability mapping

Despite the problems outlined above, and given the pressing needs of flood risk policymakers and managers, it is inevitable that impact models will be used in developing catchment flood risk management plans. As noted above, a portfolio of interventions will need to be considered, including land use management measures. 'Vulnerability mapping' represents a useful way of assembling the available knowledge and understanding into a form suitable to support decision-making. For example, it would be useful to have a map showing locations in the catchment where changes of various types should be restricted because they increase the flood hazard downstream, and conversely it would be useful to have maps that show the optimum locations to implement mitigation measures that decrease the flood hazard downstream. It would be simple to create vulnerability maps of this type if any given intervention (i.e. implemented change) is always detrimental or beneficial in the same way and by the same amount at any scale. If this was the case, then vulnerability maps could be created by agricultural scientists, soil scientists and field hydrologists, using only the small-scale information they routinely work with and map. Reality, however, is much more complicated, because a catchment is an interconnected dynamic system, so the effect of any given intervention is likely to vary from storm to storm because of changes in the spatial pattern and intensity of rainfall, changes in interactions and timings of flood waves in the drainage network, and changes in any number of other things that control runoff and routing at different scales (e.g. Burt and Slattery 1996). Conceivably, an intervention that is thought to be beneficial on initial inspection, or following short-term monitoring, might actually increase the flood hazard downstream (e.g. it might increase the 50-year return period flood).

Essentially, vulnerability maps show the sensitivity of large-scale impacts to changes in small-scale properties. It is therefore important to return to the broad question of how detailed knowledge and understanding gained at a small scale finally ends up affecting predictions made for impacts at a large scale. To create a good vulnerability map, it is necessary to track the propagation of information all the way from small-scale information (e.g. behaviour seen on field plots), through the catchment and up to the large-scale impacts, and then back again to the small scale where the (derived) data on vulnerability are plotted on the vulnerability map. Since we do not have the means available to do this tracking through the catchment itself, particularly for design floods, we have to resort to using a catchment model of suitable form.

One step that can be taken to investigate this problem is to track sensitivity as it propagates through the modelling process. This is possible using a technique called reverse (or adjoint) algorithmic differentiations (Griewank 2000; Hascoët and Pascual 2004), which can be applied to any model. It can also, as is needed here, be applied to an entire modelling process, for example from field plot measurements to flood frequency curve prediction, provided the process is fully specified as an algorithm (i.e. a set of unambiguous sequential instructions). Figure 2.8 shows a map of sensitivity derived using an adjoint version of the physically based spatially distributed catchment rainfall-runoff model SHETRAN, created during a trial of this technique (Ewen, personal communication 2009; SHETRAN 2009). For a single storm

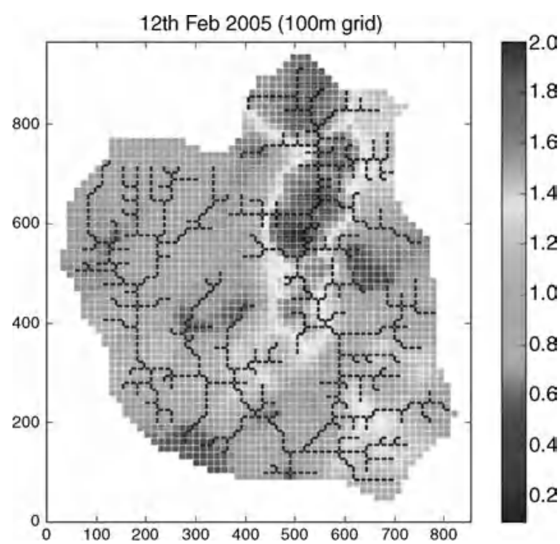

Fig. 2.8 Vulnerability map for Dunsop catchment (26 km^2) created using adjoint modelling, showing sensitivity of flood peak flow to change in Strickler coefficient. (See the colour version of this figure in Colour Plate section).

(12 February 2005), it shows the rate of change in peak discharge with a change in local-scale surface roughness (Strickler coefficient) for the Dunsop catchment, northwest England. The spatial pattern that can be seen in the vulnerability map is primarily the result of the combined effects of the spatial patterns for rainfall, soils and land cover/management, plus the effects of topography and the travel times from runoff sites to the catchment outlet. The value plotted for each grid square is for a patch of land covering 5% of the catchment area (nearest 5% to the centre of the square).

Sensitivities are, in effect, linearized responses to change, so although they can be used to estimate whether, say, a stream bank will be overtopped as a result of a change in land use/management, they cannot be used to estimate the consequences of overtopping. Other tracking methods may be useful in studying the effects of non-linearity, including methods based on unconventional types of numerical modelling that involve atomizing and tracking mass and information (e.g. Ewen 2000). When using conventional models, some form of particle tracking can be used, with information running piggyback on the particles, or packets of water can be labelled and tracked (O'Connell *et al*. 2007; O'Donnell 2008). Tracking is straightforward in simple models that use linear routing. In the geographical information system (GIS) model of Liu *et al*. (2003), for example, the outlet hydrograph is calculated by superposition as a sum of contributions from each cell, so the spatial decomposition of the hydrograph is trivial. Because sensitivity and vulnerability maps are simply forms of spatial decomposition of impact, their generation would also be trivial. The challenge with all these tracking approaches is in proving they track information well enough such that the resulting maps are useful. In a drainage network, for example, some information moves with the water molecule velocity, some with the mean bulk velocity, and some with the kinematic wave velocity, and these velocities do not behave in a simple manner even when the geometry of the channels and network is simple (Henderson 1966).

A way forward

If catchment modelling is to play its full role within a DPSIR approach, the main requirement is some form of Source-Pathway-Receptor (SPR) model that links small-scale interventions in land use/management (source) to large-scale impacts downstream (receptor), via the drainage network (pathway). In the DPSIR, this SPR model will provide the link between impacts, responses, and state. Through the use of tracking, the results from the SPR modelling can be presented and interpreted in a form that is immediately suitable as the basis for decision-making (e.g. vulnerability maps).

A suitable SPR model, with integrated tracking, is being developed in the modelling projects shown in Figure 2.4, and includes methods for estimating and tracking uncertainty. The main elements of the SPR model are the metamodelling being developed by Imperial College (see detailed description in Chapter 3) and the tracking being developed at Newcastle University. When forced with synthetic rainfall representing present or future climatic conditions, the SPR model produces vulnerability maps that show the link between the flood frequency curve and potential/proposed interventions in land use management. These maps can then be used within the other components of a DPSIR system, as a basis for decision support. The tracking uses a detailed drainage network model (currently, a non-inertia St Venant approach) allied with impact sensitivity tracking using adjoint modelling. The development and testing of the SPR modelling is relying heavily on the multiscale field experiments shown in Figure 2.4 and an existing rainfall generation model (Burton *et al*. 2008).

Discussion and Conclusions

The UK's rural landscape has been undergoing continuous change for more than 50 years, and will continue to undergo change in the future in response to climatic and socioeconomic drivers. Moreover, climatic change is expected to enhance flood hazard in the coming decades. A broad portfolio of flood risk management measures will need to be considered as part of any catchment flood risk management plan, in line with the MSW strategy, and the environmental, social and economic dimensions of sustainability will need to be assessed for any portfolio of measures. Moreover, the measures implemented will need to be adaptive and resilient under climate change pressures.

A strategic approach to the analysis and modelling of change necessitates the adoption of a broad holistic modelling framework that encompasses the climatic and social drivers, the prediction of impacts and the integrated assessment of response measures based on economic, social and environmental criteria. The DPSIR framework for the analysis of change has been integrated with the Source-Pathway-Receptor (SPR) modelling of impacts and responses to provide a strategic approach that can support sustainable flood risk management. In implementing such an approach, the major challenges are to integrate all of the technical elements into the SPR modelling approach, and then to incorporate the SPR modelling into an integrated sustainability assessment. The latter will involve confronting the formidable challenge of integrating the technical and social aspects of flood risk management that have heretofore tended to be researched within separate disciplinary domains, but which are encompassed by FRMRC research.

This chapter has focused on the technical subject of how land use management interventions can impact runoff generation at source and possibly downstream flooding within a catchment, and has considered how any adverse impacts might be mitigated. Catchment-scale modelling of the rainfall runoff process is at the core of the SPR modelling, with the source being at the scale of the farmer's field, the pathway being the drainage/river channel network, and the receptor being the point of downstream impact. Current rainfall-runoff models are not fit for purpose in this regard, and new concepts such as information tracking and adjoint modelling are being developed as key elements of SPR modelling and the construction of vulnerability maps. The SPR modelling approach links local-scale metamodels that emulate detailed physics-based models of runoff generation, the regionalization of the metamodel parameters, and their linkage to a fine-scale channel network routing model. The upper limit on the scale at which this approach can operate is typically of the order of hundreds of square kilometres, and is defined by the scale at which floodplain inundation starts to have a significant influence on the flood hydrograph, Therefore, in order to assess impacts on an urban receptor, for example, the mesoscale catchment models would need to be linked with the flood inundation models described elsewhere in this book (see Part 4, Flood Modelling and Mitigation).

The validation of the SPR modelling approach and the estimation of uncertainty in model predictions is a subject of critical importance in a decision-making context, and is the subject of Chapter 14 by Beven. This subject has not been discussed in any detail above, but the SPR modelling will employ state-of-the-art approaches to the conditioning of model predictions using observed data and the estimation of prediction uncertainty. The information tracking approach can be used to track uncertainty as it propagates through the model chain, and it is therefore intended to map the uncertainty in the vulnerability maps.

In an ideal world, a book of this kind would contain a prescription for a methodology that would allow land use management impacts to be assessed, and appropriate flood risk mitigation measures to be evaluated and put in place within the landscape as a routine procedure. Unfortunately, we are still some way from achieving this ideal, but are making substantial progress in constructing the key building blocks for such a methodology. At this point in time, the following conclusions can be drawn:

1 There is substantial evidence that modern land use management practices have enhanced surface runoff generation and flooding at the local scale.

2 No clear evidence has yet been found that changes in land use management practices have created impacts at the catchment scale. This may be because of the overriding effects of climatic variability, and because high-quality data that would allow impacts to be uncovered are not generally available.

3 As a consequence of point 2, the impact at catchment scale of a range of local-scale mitigation measures distributed across the catchment landscape is unknown, and predictions of impact must await the outcomes of field experiments and new model developments.

4 Multiscale experimental programmes are currently in place, which are providing high-quality multiscale data on the impacts of land use management changes (including mitigation measures) on runoff generation and flooding.

5 A broad holistic modelling framework is needed that encompasses the climatic and social drivers, the prediction of impacts and the integrated assessment of response measures based on economic, social and environmental criteria. The combination of DPSIR (Drivers-Pressures-State-Impacts-Responses) analysis with broad-scale modelling (BSM) provides such a framework, but its development must be supported by a suitable research programme. A programme being followed in the UK has been described, which includes an integrated programme of multiscale field monitoring/experimentation and modelling, including an analysis of uncertainty.

6 The modelling and prediction of impacts using catchment models is fraught with a number of major difficulties that inhibit the prescription of a straightforward modelling approach; these difficulties are being addressed within a current FREE/FRMRC2 research programme. Model users should be aware of the (serious) limitations of the current models.

7 The preferred framework for predicting impacts involves linking metamodels of local-scale changes to a fine-resolution network routing model for propagating impacts to larger scales.

8 New techniques for Source-Pathway-Receptor (SPR) modelling and the construction of vulnerability maps, for use in DPSIR analysis of potential/proposed interventions, are being developed based on information tracking and adjoint modelling concepts (the vulnerability maps show in detail the link between small-scale interventions and large-scale downstream impact).

9 Any strategic approach to catchment flood risk management must take explicit account of the drivers that may alter land use and management in the future, with consequences for future flood risk. This necessitates the adoption of a broad holistic modelling framework such as DPSIR-BSM that encompasses the climatic and social drivers, the prediction of impacts, and the integrated assessment of response measures based on economic, social and environmental sustainability criteria.

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3 Multiscale Impacts of Land Management on Flooding

HOWARD S. WHEATER, NEIL MCINTYRE, BETHANNA M. JACKSON, MILES R. MARSHALL, CAROLINE BALLARD, NATALIYA S. BULYGINA, BRIAN REYNOLDS AND ZOE FROGBROOK

Introduction

Land management change

Over the last 50 years there have been significant changes in the rural landscape of the UK as a result of agricultural intensification. Obvious landscape features include the removal of hedgerows, increased field size and changing cropping patterns. In the lowlands, arable agriculture has changed, with a move to autumn-sown crops, and this has led to bare soil conditions over autumn and winter. In addition, management change has led to the use of contractors and larger machinery for farm operations, and hence heavy equipment operating on the land with less opportunity to consider appropriate weather conditions for field operations. In the uplands, pasture has been 'improved', by installation of agricultural drainage, ploughing, reseeding and fertilization, and grazing densities have increased dramatically, also leading to pressures to use less suitable land under unsuitable soil conditions. There is widespread anecdotal evidence in the UK that these pressures on agriculture have led to degradation of soil structure, through a combination of surface capping of arable soils, poaching of grassland soils, and topsoil compaction in both (Holman *et al*. 2002; Wheater 2002; O'Connell *et al*. 2004, 2007). This is not an issue solely confined to the UK; similar concerns have been raised elsewhere across northern Europe (Savenije 1995; Bronstert *et al*. 2002; Pfister *et al*. 2004; Pinter *et al*. 2006; Evrard *et al*. 2007).

It is thought that agricultural intensification may cause higher flood peaks in streams and rivers due to the impact on runoff processes. For example, degradation of soil structure can lead to reduction in soil infiltration rates and available storage capacities, increasing rapid runoff in the form of overland flow (Heathwaite *et al*. 1990; Bronstert *et al*. 2002; Holman *et al*. 2003; Carroll *et al*. 2004b; O'Connell *et al*. 2004). This may increase the risk of flooding (Stevens *et al*. 2002; Holman *et al*. 2003), as reported both in the UK and across northern Europe, particularly on intensively farmed soils (Boardman *et al*. 1994; Burt 2001). Although hard scientific evidence on impacts is limited, there is information on the extent of soil degradation. For example, a survey during the autumn 2000 floods (Holman *et al*. 2002) concluded that 30% of soil in the Severn catchment was degraded and 55% of areas with late-harvested crops showed severe soil degradation. Such data reinforce concerns for localized flooding (so-called muddy floods) and the possibility that such changes, if of sufficient spatial extent within a catchment, may significantly alter the hydrology of major rivers.

If effects of land management are significant in influencing hydrological response at the

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catchment scale, that is clearly important in terms of flood risk assessment, but also raises the possibility that appropriate land management interventions might be adopted to reduce downstream flood risk. Interventions with potential benefits for flooding may also be relevant to other aspects of environmental management, such as the control of diffuse pollution and habitat improvement to support improved biodiversity. There has been increasing recognition in recent years from policy-makers of the interdependence between land use and water management. For example, in the UK, current policy on flood risk management, in particular Defra's 'Making Space for Water' (MSW), recognizes that water management is inextricably linked to land management. There is therefore an urgent need for guidance concerning the hydrological impacts of land management to inform agricultural policy.

As a result of the above, land use and management as a source of flooding was one of the key research priority areas identified by the Defra/Environment Agency (EA) R&D programme (Calver and Wheater 2002; Wheater 2002), and subsequently by the Engineering and Physical Sciences Research Council (EPSRC), following extensive consultation, in establishing the UK's Flood Risk Management Research Consortium (FRMRC). A major review was commissioned by Defra (O'Connell *et al.* 2004), which concluded that the role of land use management in enhancing and/or ameliorating UK flood risk was an unanswered question. And although the risk of flooding is concentrated in lowland regions, the management of catchment headwaters, with their generally higher precipitation rates and flashier response, is of particular interest for flood runoff generation. To address this issue, the FRMRC made a major investment in a multiscale experimental and modelling programme, based at Pontbren in mid-Wales. In this chapter we report on the results of FRMRC Phase 1 (FRMRC1), and introduce preliminary results from the continuing research in this area, funded under the FRMRC Phase2 and the Natural Environment Research Council's (NERC's) Flood Risk from Extreme Events (FREE) research programme.

Quantification of hydrological impacts of land management change

As discussed above, there is an urgent need to quantify the hydrological effects of land use and land management change, and clearly the potential for reversal is also important, but available guidance to represent land management effects is limited at best.

As noted above, O'Connell *et al.* (2004) carried out an extensive review, which highlighted the lack of evidence for local-scale and, particularly, catchment-scale effects. As an interim measure, they proposed a speculative modification to the Hydrology of Soil Types (HOST) classification of UK soils to represent the potential effects of soil degradation on runoff production. JBA (2007) applied the methodology to the Ripon catchment. Results indicated that, if soil structural degradation were to occur across the whole catchment, together with additional maintenance of moorland grips, peak flows in the town of Ripon would increase by between 20% for smaller scale floods and 10% for more extreme floods. A less extreme scenario (soil degradation over 30% of the catchment) led to increased peak flows of 10% for smaller scale floods and 3% for more extreme events. While these results appear entirely plausible, they are essentially speculative. However, the methodology is attractive, and provides the basis for a simple method with national applicability. Two important issues arise. First there is a lack of an evidence base to support the HOST class modifications at either local or catchment scale; and second, there is potentially a wide range of interventions that can be considered at local scale, such as the planting of tree shelter belts and the development of farm ponds. Such interventions could not be directly related to a notional level of soil degradation.

One option to identify the catchment-scale effects of land use and land management change is to interrogate catchment-scale data, and this has been pursued recently by Beven *et al.* (2008), using available UK datasets. The study failed to identify a clear relationship between land use/management and river flows. However, it is important to note that this does not mean that such

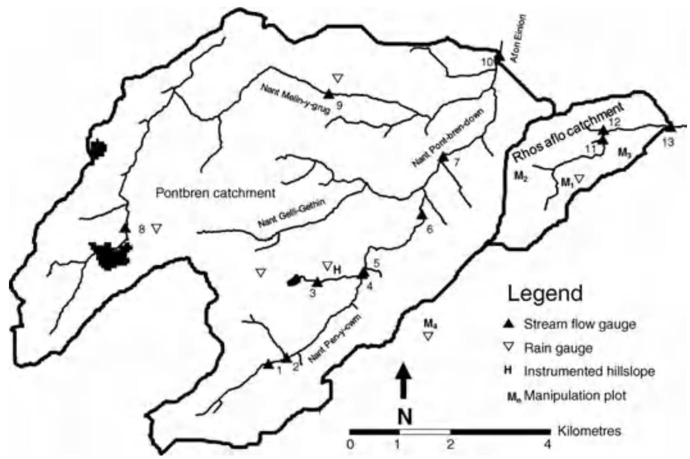

Fig. 3.1 Pontbren and Rhos aflo catchment showing the main streams and gauging sites.

a relationship does not exist, but rather that noise in catchment-scale measurements and the multifaceted nature of catchment change, combined with climate variability, do not allow such effects, even if potentially substantial, to be discriminated.

There is therefore a need for an alternative approach. There are two basic issues that arise. The first is the availability of appropriate data and models to characterize local-scale effects; and the second is the development of an appropriate modelling strategy to address the generic problem of upscaling from local to catchment-scale. In this chapter we report the development and application of such a methodology, based on multiscale experimental data from Pontbren, a tributary of the Severn in mid-Wales, and discuss the potential of this approach for extension to national application, linking back to the use of regional datasets, such as the HOST classification, and the simplified methods of O'Connell *et al*. (2004).

The Pontbren Multiscale Experiment

Background and experimental design

Pontbren, situated in the headwaters of the River Severn in Wales, is a farmers' cooperative concerned with sustainable upland agriculture, involving 10 hill farms and over 1000 ha of agriculturally improved pasture (drained, ploughed, reseeded and fertilized) and woodland (Fig. 3.1). Elevations range from 170 to 438 m AOD (Above Ordnance Datum), and the soils are clay-rich, mainly from the Cegin and Wilcocks series, which are common in Wales. The soils have low-permeability subsoil, overlying glacial drift deposits, and are seasonally wet or waterlogged. Field drainage is ubiquitous where pasture has been improved. The predominant land use is grazing, mainly for sheep.

The farmers' perception is that changes to land management, and in particular changes to grazing densities and animal weights, have changed runoff response. Although land use has changed relatively little since the 19th century, between the 1970s and 1990s dramatic changes in farming intensity took place; sheep numbers increased by a factor of six and animal weights doubled (R. Jukes, personal communication).

The farmers have recently reinstated woodland areas and hedgerows, and preliminary research by the Centre for Ecology and Hydrology, Bangor (Carroll *et al*. 2004) on the infiltration rates of the grazed hillslopes and woodland buffer strips demonstrated a dramatic change in soil response to rainfall. Infiltration rates on the grazed pastures were extremely low, but within a few years of tree planting, soil structure and permeability in buffer strips showed significant improvement. However, these results needed to be extended, to evaluate land use impacts in a statistically rigorous manner and to address runoff generation at the hillslope and catchment scale. The effects must also be evaluated in the context of the complex history of land management. For example, land drainage is extensive, with a history that is believed to go back to the Napoleonic Wars

Multiscale experimentation is needed to bridge the current gap between plot-scale experiments and catchment-scale impacts, hence a set of experiments was designed to provide data support for new methodological development. The Pontbren project crucially provides landowner support for land access, land management manipulation experiments and for socioeconomic analysis. The direct involvement of the local farmers is also important in the promotion of policy guidance to the agricultural community.

The current scales of research range from experimental plots to an 18-km^2 catchment, including three first-order streams. The experiments focus on soil properties and runoff processes, based on plot- and hillslope-scale measurements nested within instrumented first- and second-order catchments (Marshall *et al.* 2006). At plot scale, manipulation plots have been established at four locations, representing a range of aspect and soil type. At each location three treatments are being evaluated: grazing, no grazing and newly planted woodland. Continuous monitoring includes precipitation, other climate variables, soil moisture contents, soil water potentials and overland flow. In addition, soil physical and chemical properties are characterized in annual sampling campaigns. At hillslope scale, instrumented hillslope transects include the above instrumentation, groundwater elevations, and drain and ditch flows. Within the hillslope experiments, soil properties and runoff processes are being investigated under different land use treatments including woodland buffer strips. At catchment scale the monitoring is complemented by a network of stream gauges. These observations are supported by a soil survey, including estimation of soil degradation status, supplementary sampling and additional experimentation, including sprinkler and tracer experiments and woodland interception studies. The data are extensive; approximately 145,000 data items are being recorded per week.

Experimental results

Figure 3.1 shows the Pontbren and adjacent Rhos aflo catchment along with the main streams within the catchment and monitoring locations. A large dataset now exists from the Pontbren experiment, and results are presented here to illustrate key findings. These include the impact of plot-scale land use change on soil hydraulic properties and hydrological processes, runoff processes from an improved pasture hillslope, and stream flow response. More extensive results are provided by Wheater *et al.* (2008).

A survey assessing the structural conditions of the soils at Pontbren was undertaken in 2006 and followed the methodology described in Holman *et al.* (2003). The survey indicated that 60% of the land was moderately degraded and 25% of the land was highly degraded, and that these highly degraded soils were confined to land under improved grassland production. There is also anecdotal evidence to indicate that in recent years the Wilcocks soil series, with its peaty surface layer, may have retreated up the hillslope, being replaced by the Cegin soil series (a heavy-textured clay loam). It is speculated that this loss of peat is attributed, to some extent, to agricultural intensification, such as the increase in stock numbers and the installation of field drainage systems. The improved pasture tends to dominate in the east of the study site whereas the rough grazing is found more to the west and north on the higher altitudes.

Manipulation plots

The response of the manipulation plots is highly heterogeneous, hence it was important to establish baseline conditions prior to the imposition of the experimental treatments. The plots were therefore established in 2006, and treatments implemented in 2007. The monitoring and analysis is ongoing, but despite the treatments being less than 2 years old, changes in soil physical and biological properties have been observed. For both treatments, i.e. no grazing and planting of trees, a significant (t-test significant at the 0.05 level for both treatments) reduction in soil bulk density for the top 3–5 cm of the soil has been measured. There is also a significant increase in the number of earthworms observed within the top 15-cm depth of soil under both treatments (ANOVA returned an F-value of 0.001 for treatments). The presence of earthworms can have a significant impact on the movement of water through soil (Li and Ghodrati 1995), especially in heavy-textured soils such as those found at Pontbren.

Changes in hydrological response have been observed post-treatment. Relative increases in soil infiltration rate and a reduction in overland flow occurring in treatment plots compared to the grazing control are observed at three out of the four plots. Figure 3.2 illustrates an example of this from manipulation plot M_2. For the one site where no apparent difference between treatment and control has yet been detected, conditions are particularly wet, due to the heavy-textured soil and the slope location. This research is ongoing, including quantitative analysis of soil properties, soil states and runoff processes.

Hillslope monitoring

In considering the catchment-scale response it is important to understand the nature of the runoff processes at hillslope scale; hence, detailed monitoring of climate, soil water states and runoff processes was established for an instrumented hillslope ($\sim$ 0.4 ha) under improved grassland production. The results illustrate the importance of drain and overland flow (OLF) in characterizing the runoff response at this scale of observation. Figure 3.3 indicates that the field drainage systems, which are extensively installed throughout the areas of improved grassland, appear to dominate the runoff response in terms of volume at the hillslope scale. However, there are times when overland flow rate exceeds that of the drain flow and is a significant contributor to total runoff peak flow rates. Pore water pressure readings indicate that when overland flow occurs it does so as a result of saturation excess and not infiltration excess. This is illustrated in Figure 3.4, where there is relatively more overland flow occurring during the first event (a) compared to the second event (b) due to the saturated soil conditions indicated by the pore water pressure data in corresponding plots below. A relatively impermeable

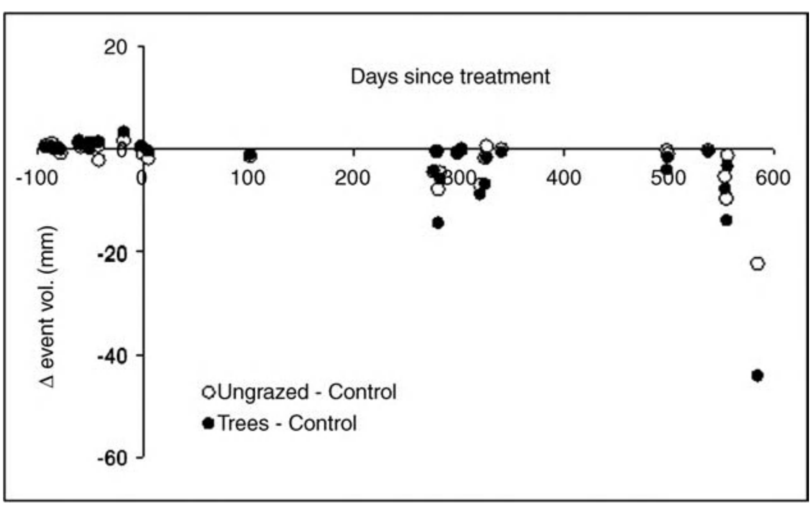

Fig. 3.2 Difference in overland flow runoff event volume (treatment – control) (mm) from manipulation plot M_2.

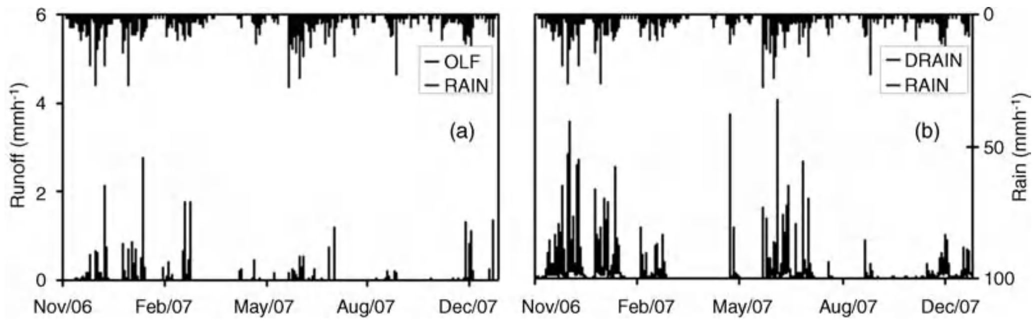

Fig. 3.3 Standardized overland flow (OLF) and drain flow (DRAIN) from the instrumented hillslope for the period November 2006 to December 2007 (data from Marshall *et al.* 2009).

subsurface layer prevents much downward movement of incoming rainfall, resulting in a near-surface perched water table over an essentially hydrologically isolated groundwater system that is normally relatively unresponsive. Dye tracer studies have illustrated the importance of preferential flow pathways in promoting the rapid movement of water down through the soil profile and into the field drainage systems.

During $3^1/_2$ years of groundwater monitoring the only time when the groundwater became responsive to incident rainfall was in the winter of 2006/7, following an exceptionally hot, dry summer. Preferential flow paths in the form of inter-pedal cracks developed in the clay-rich soil, which allowed water to bypass not only the A horizon soil matrix but the normally impermeable subsurface layer. The development of such extensive preferential flow paths also resulted in a change in runoff with an increase in responsiveness of the drain flow. However, the dynamic nature of the bulk soil physical properties and resultant effect on runoff is not confined to extreme climatic conditions and appears to be a continuous annual cycle. The long-term dataset from the instrumented hillslope has indicated seasonal changes in runoff response. An increase in the flashy nature of the drain flow during the summer and into the autumn is

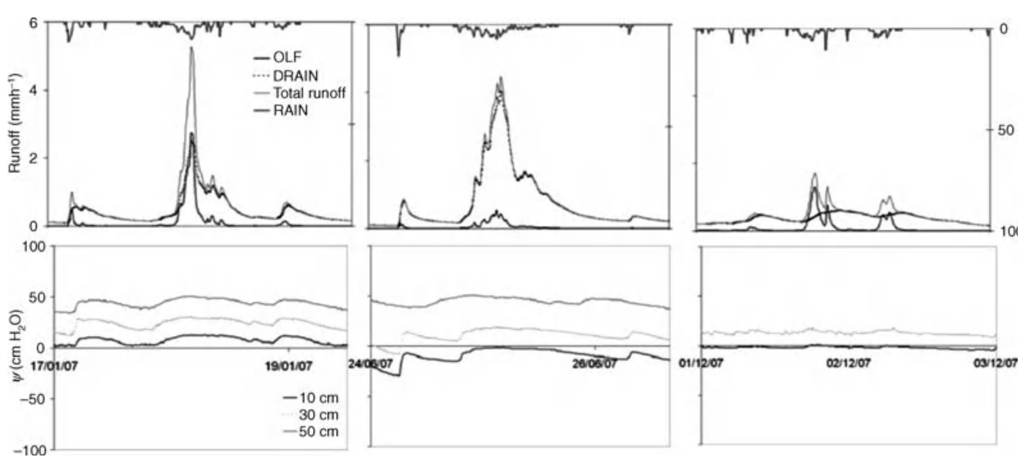

Fig. 3.4 Bowl runoff data, and corresponding pore water pressure data, ψ (cmH_2O) (plots a, c, e), from tensiometers installed at 10, 30, and 50 cm (plots b, d, f) (50 cm depth tensiometer not in operation during 01–02/12/07 event) (data from Marshall *et al.* 2009).

observed, possibly as a result of the increase in development of preferential flow paths in the spring and summer caused by the soil cracking and development of earthworm channels. During winter, a gradual decrease in the effectiveness of these preferential flow pathways is observed, presumably as they begin to deteriorate. This results in an increase in the relative importance of overland flow. This is illustrated by comparing the difference in drain flow response for the three events shown in Figure 3.4. The results demonstrate the need for long-term monitoring in order to capture the annual and interannual changes in runoff response as result of climatic variability.

Differences in soil properties are observed at the instrumented hillslope comparing soil under improved pasture and that under an established tree shelterbelt. A significant increase in the saturated hydraulic conductivity is measured in the A horizon of soil under trees compared with that of the same soil type under improved pasture. There is also a change in the soil moisture release curve, with the soil under trees showing an increase in available pore space between saturation and field capacity ($\Psi = -100\,\mathrm{cmH_2O}$; Rowell 1994). There was, however, little difference in the B horizon between land uses. Carroll *et al.* (2004) found a significant increase in soil infiltration rates at Pontbren due to the presence of trees and/or the absence of sheep. Comparing overland flow from two 5×5 m isolated plots within the tree planted shelterbelt and that from the previously discussed improved pasture, a 71% and 66% reduction occurred in the plots under trees collected over a 10-month period (Marshall *et al.* 2009).

Catchment-scale response

A clear similarity in runoff response is observed comparing total runoff (drain flow and overland flow combined) from the instrumented hillslope under improved pasture with hydrographs measured at gauging sites on the Nant Pen y cwm stream (Fig. 3.5; see also Fig. 3.1). One might expect this to be due to the close proximity of the sites and the fact that a large proportion of the land draining to these sites is under improved grassland production (see Table 3.1). When moving from the hillslope scale to the stream flow network a decrease in flood peak (per unit area) is measured as a result of increasing variability in travel times.

However, differences in stream flow response are observed by comparing subcatchments within Pontbren where the land use is significantly different. Figure 3.6 shows the hydrograph of a significant runoff event for gauging sites 6 and 9 (see Fig. 3.1). The peak runoff rate at site 9 is much reduced compared to that of site 6. As opposed to Nant Pen y cwm, the Nant Melin y grug subcatchment has remained relatively unchanged, with much of the land still managed as unimproved

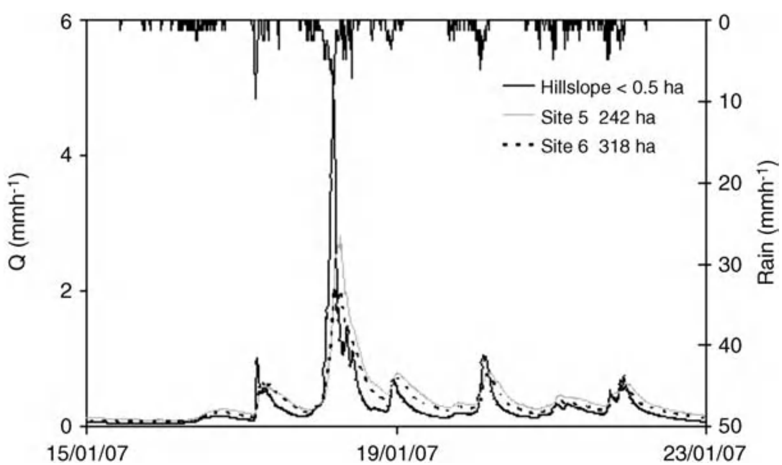

Fig. 3.5 Standardized total runoff (Q) (drain flow and OLF), from the improved pasture hillslope and stream flow response from gauging sites 5 and 6.

Table 3.1 Pontbren subcatchment characteristics

Site	Area (ha)	Mean slope angle (SD) (degrees)	Area under improved pasture[a] (%)	HOST class[b]	SPR[c] (%)
Hillslope	<0.5	7.3 (0.57)	100	21.8	41.9
5	242	4.74 (2.93)	70	24.6	52.5
6	318	5.04 (3.08)	76	23.9	51.0
9	402	6.95 (6.36)	14	19.4	48.4

Source: Countryside Council for Wales phase 1 survey data.
[a] Hydrology Of Soil Type Classification (Boorman *et al.* 1995).
[b] Standard Percentage Runoff.
[c] SD, standard deviation.

pasture and open moorland (see Table 3.1). There is little difference in geomorphology between catchments, and the potential dampening response of the lake at the source of the Melin y grug appears to have relatively little effect on flood peak. There are differences in the soil types. Table 3.1 shows the spatially averaged HOST (Hydrology Of Soil Type) classification (Boorman *et al.* 1995) for each of the subcatchments along with the estimated standard percentage runoff (SPR) based on the HOST class value. Results indicate that differences in runoff response may arise from a combination of differences in soil type and land use, but the relative contributions are difficult to disentangle.

In summary, the experimental results have clearly demonstrated the dominant hillslope runoff processes, and provided insight into the time-varying nature of responses as soils respond to seasonal wetting and drying and increases in biological activity. The plot experiments are ongoing, but relatively rapid changes have been observed in soil structure and biodiversity. Observations of established tree shelter belts show significant changes to soil hydraulic properties, with increased infiltration capacity. Results are presented in more detail in Wheater *et al.* (2008).

Multiscale Modelling

Modelling strategy

The modelling strategy has three elements. At Pontbren we are concerned with representing physical changes to soil structure, vegetation and field drainage, and the associated effects on runoff processes. A key element therefore is the establishment of a detailed, physically based model, capable

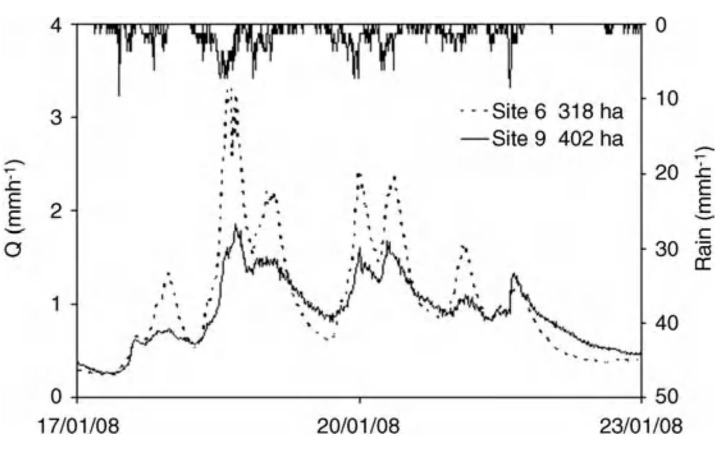

Fig. 3.6 Standardized stream flow response (Q) from gauging sites 6 and 9.

of representing the important hydrological processes operating at Pontbren and similar catchments, at the scale of individual fields and hillslopes. For this we have developed further an Imperial College model based on Richards' equation for saturated/unsaturated soil water flow (Karavokyris *et al*. 1990), now extended to represent macropore processes and overland flow, incorporating vegetation processes (such as interception) and associated effects such as changing root depths and soil hydraulic properties, and capable of being run in one, two or three dimensions (Jackson *et al*. 2008). The model has been conditioned, within a Monte Carlo-based framework of uncertainty analysis, using physically determined soil hydraulic properties and continuous measurements of climate inputs, soil water states and runoff (as overland flow and drain flow) from the Pontbren experimental sites. Due to the highly non-linear dynamics, individual fields and hillslopes are represented at fine resolution (1 cm vertical and 1 m horizontal resolution). The detailed model can be exercised to simulate scenarios of interest, including, for example, the planting of strips of woodland within a hillslope, and the associated changes to soil structure, evaporation processes, overland flow and drainage.

The detailed model is computationally intensive, and not suitable for direct application at catchment scale. The second element of our strategy is therefore to use a metamodelling procedure, whereby the detailed model is used to train a simpler, conceptual model that represents the response in a parsimonious and computationally efficient manner, using basic hydrological components of loss and routing functions. This requires classification of the landscape into hydrological units, based, for example, on soils, land use and existing/proposed interventions. The detailed model is run for each member of a library of hydrological units, and hence a metamodel parameterization is obtained for each member through the model training process. Uncertainty in parameter values is carried forward to this stage via Monte Carlo analysis.

The third element of the procedure is a catchment-scale semi-distributed model, written with a modular structure, which uses the metamodel elements to represent individual hydrological elements, and routes the flows down the stream network. Using the semi-distributed model, the metamodel is further conditioned on the catchment-scale data to reduce parameter uncertainty.

Illustrative results are presented below; full details of the data and modelling are presented in Wheater *et al*. (2008).

Detailed modelling of local-scale effects

In order to represent the hydrological processes at Pontbren, a model (or models) needs to be created. The model comprises two parts: a perceptual model of processes, which is an attempt to describe the key processes that are perceived to be occurring; and a numerical model, which is a mathematical implementation of the perceptual model. The development of a perceptual model is influenced by the available information, which can include topography, soil classification, 'soft' observations made in the field (e.g. presence or absence of cracks in the soil), processes implied from the evaluation of data, and direct experimental observations such as state observations, flux data and chemical or dye tracer tests.

Data from the Pontbren hillslope study site (the bowl) suggest that, if representative of improved grassland over the catchment generally, improved grassland has the following hydrological characteristics:

- Over much of the winter there is a perched water table caused by the low permeability of the soil B horizon in wet conditions.
- Groundwater is generally dissociated from incident rainfall (with a notable exception at the end of summer 2006).
- Soil moisture shows negligible change below 70 cm (drain level).
- There is evidence of macropore flow at small scales; however, the presence of a perched water table over much of winter suggests that there is either limited connectivity of these macropores, or they exist in limited regions. They appear to be less important in the B horizon than in the A horizon.
- Flood generation in winter conditions is often dominated by overland flow.

• Drain flow remains important in winter flood conditions, and may be the dominant flow mechanism in most summer floods.

There is certainly significant small-scale heterogeneity and strong evidence for significant non--stationarity of small-scale responses. There is very little understanding of compaction effects in these soils, to allow us to distinguish the response of grazed and ungrazed grassland.

Data from tree-planted areas in Pontbren suggest:

• increased infiltration within tree shelter belts;
• increased capacity to store water underneath trees;
• large interception losses;
• significant tree shelter-belt edge effects.

Despite the extensive field programme, some uncertainties in the perceptual model remain. A major knowledge gap regarding the tree-planted areas is the fate of subsurface water. For extreme events, the interception and localized near-surface storage that is known to be associated with the trees is not necessarily adequate to significantly reduce flood generation; and the activation of slow subsurface pathways and deeper stores becomes important. More knowledge about the subsurface routes below the trees is therefore desirable. We assume here that connection to field drainage systems exists.

We also have incomplete understanding of the relative functioning of unimproved grassland and wetland areas within Pontbren. Hence these areas are currently modelled and conditioned using catchment-scale data: data show that these areas are less flashy than improved grassland; however, the extent to which the unimproved grassland dampens response is an open and important question currently being addressed.

Figure 3.7 illustrates the simulated response for a representative hillslope (100×100 m) using the detailed model for a range of land management types, including grazed and ungrazed drained grassland, grassland with tree shelter belts (80 m length, 15 m width) in different locations, and full tree cover. The envelopes of response represent the range of parameter uncertainty.

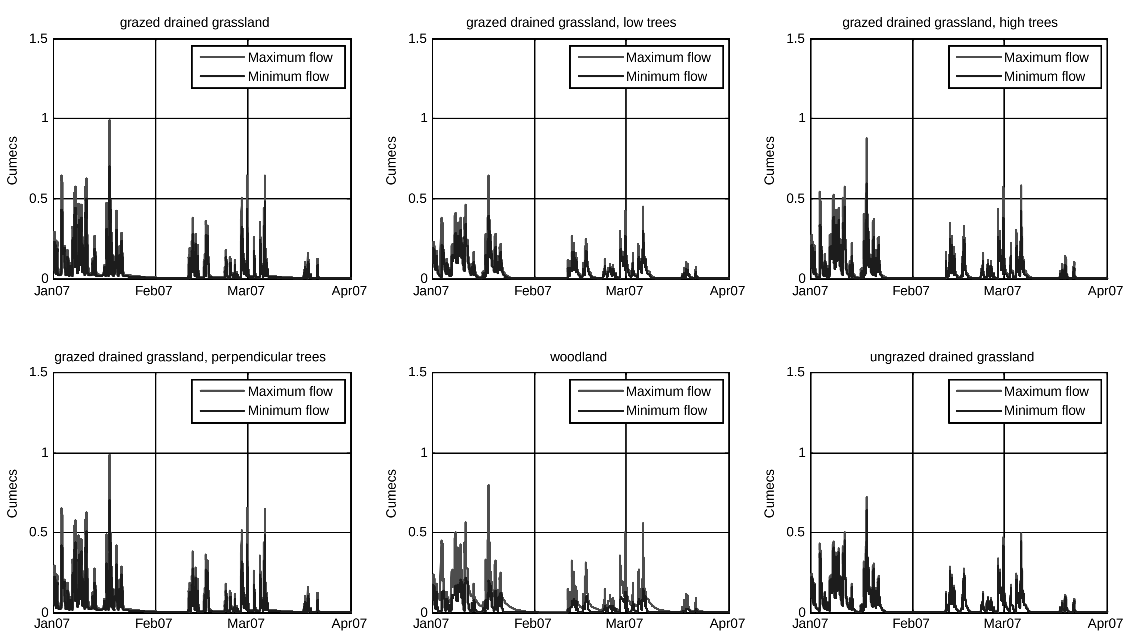

Fig. 3.7 Field-scale runoff (drain flow + overland flow) for different land use types, with uncertainty bounds.

Metamodel performance

The aim of the metamodel is to provide a computationally efficient model that is able to capture the response of the detailed physics-based model; ideally the model would retain some physical interpretability through the use of an appropriate conceptual structure. A common simple representation of the rainfall-runoff process is to consider a loss function, representing soil water controls on evaporation, and routing, typically representing fast and slow response pathways – the Imperial College Rainfall-Runoff Modelling Toolbox (RRMT; Wagener *et al.* 2004) has a large library of alternative structures. A suitable model structure was defined (Fig. 3.8).

The metamodelling strategy requires that each field in the Pontbren catchment is classified into a land use/management type, so that the corresponding set of field-scale models can be applied. The field types currently included are:

1 grazed improved grassland;
2 tree belt/hedgerow: near bottom of slope;
3 tree belt/hedgerow: near top of slope;
4 tree belt/hedgerow: 90° to contour;
5 woodland;
6 ungrazed improved grassland;
7 grassland with drains removed;
8 unimproved grassland/rough grazing;
9 marsh/wetland.

These units were chosen based on dominant land use types currently within the catchment and those management changes that were perceived as likely to have an impact on flood peaks.

The ability of the metamodel to represent detailed model response is illustrated in Figure 3.9 for a grazed hillslope with a woodland buffer strip at the base of the slope (category 2 above).

Those metamodels that are conditioned on physics-based models (namely: grazed improved grassland, ungrazed improved grassland, improved grassland with no drains and three tree-belt interventions) are by implication conditioned on small-scale data. Uncertainty is handled through generating multiple samples of physics-based model responses to account for uncertainty on the data. Each individual detailed model simulation is then passed to the metamodel and a corresponding response generated through automatic calibration of the metamodel to this response, using Monte Carlo simulation and a least squares fit measure. This allows a set of behavioural samples to be propagated forward

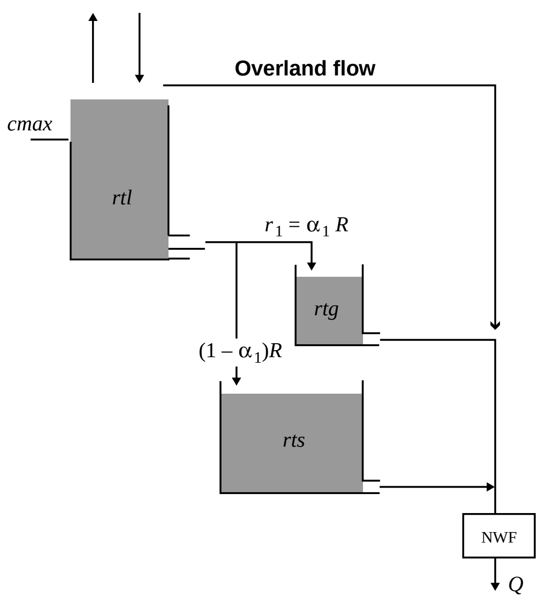

Fig. 3.8 Metamodel structure and associated parameters.

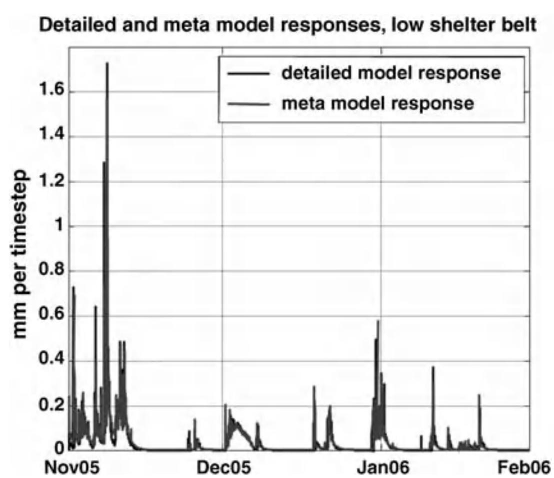

Fig. 3.9 Metamodel performance.

within the catchment model to take account of small-scale data uncertainty and metamodel uncertainty.

Catchment-scale simulations

The semi-distributed rainfall-runoff modelling toolbox RRMTSD (Orellana *et al*. 2008) is a modular framework that allows efficient building and evaluation of semi-distributed rainfall-runoff models (Fig. 3.10). These models are semi-distributed in the sense that the watershed is conceptualized as a network of sub-areas for which lumped conceptual rainfall-runoff models are computed. The hydrological processes and climatological forcing data within the sub-areas are considered to be homogeneous, and the degree of spatial distribution is represented mainly through the number of sub-areas. These can represent subcatchments or hydrological response units, and can incorporate the metamodel structures discussed above.

Topologically, RRMTSD simulates streamflow for the uppermost stream sub-areas first and then continues with the downstream ones. The architecture comprises three component modules: moisture accounting, runoff routing and channel routing. The first module determines effective rainfall (ER), actual evapotranspiration (AET) and an estimation of moisture status; the routing module calculates the fast and slow runoff; and the channel routing module estimates discharge at the outlet of the sub-area. The formulation of the first two modules is based on the established RRMT framework (Wagener *et al*. 2004). A variety of pre-built modules are available, which are interchangeable, but others can be added providing additional flexibility.

The toolbox allows for different optimization methods for calibration: uniform random search, the shuffled complex evolution method (Duan *et al*. 1993), and local non-linear multi-constrained methods based on simplex searching. These methods can be applied with the same or different model structures representing the individual sub-areas. The input data and simulated variables in every sub-area can be analysed using a variety of visualization tools.

This semi-distributed rainfall-runoff model is used to route water from field-scale hydrological response units to and along the streams and ditches identified within the catchment. Fields were chosen as the individual response units because these seem an appropriate management unit when looking at the influence of land use changes.

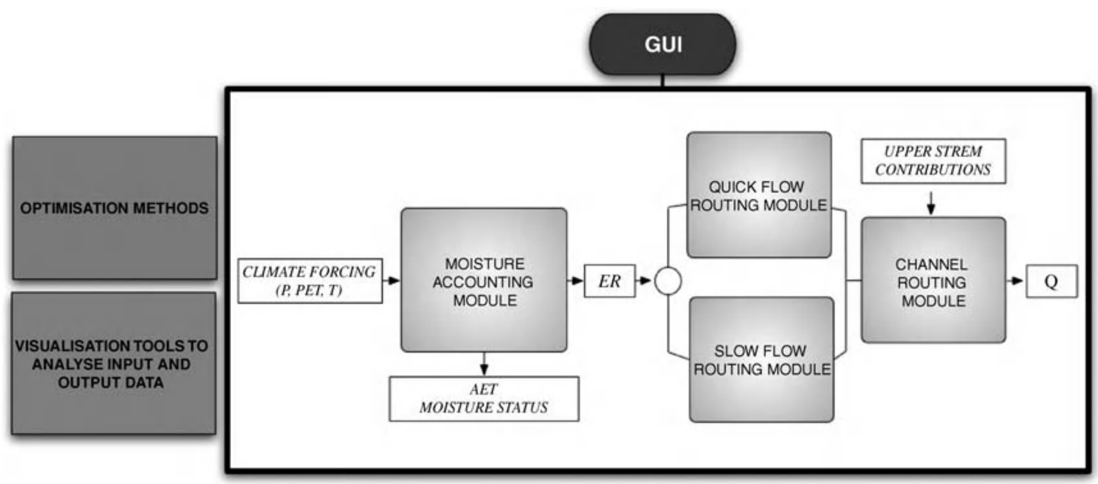

Fig. 3.10 Structure of Semi-Distributed Rainfall-Runoff Modelling Toolbox (RRMT-SD). P, precipitation; PET, potential evaporation; T, temperature; AET, actual evaporation; ER, effective rainfall; Q, discharge.

They also generally form sensible individual units, due to the tendency of farmers to set ditches and drainage outlets at field boundaries.

We illustrate the impacts at the catchment-scale in Figure 3.11, for a 4-km^2 Pontbren sub-catchment. The baseline is the current-day land use at Pontbren, the first scenario removes the effect of the recent Pontbren tree plantings (and hence takes the catchment back to something approximating the intensive use of the early 1990s), the second adds shelter belts to all grazed grassland sites, and the third assumes the entire catchment is woodland. The changes in flood peaks observed for the three scenarios are: removing all the trees causes up to 20% increase in flood peaks from the baseline condition; adding tree shelter belts to all grazed grassland sites causes up to 20% decrease in flood peaks from the baseline condition; and full aforestation causes up to 60% decrease in flood peaks from the baseline condition.

Within FRMRC1, therefore, a new modelling procedure has been defined, which allows explicit representation of detailed management changes at the scale of individual fields and hillslopes, and provides, through metamodels, the upscaling necessary to simulate catchment-scale effects. The results of Figure 3.11 show some relatively large effects for relatively frequent events. Clearly extreme events are of particular interest for flood risk assessment, and Wheater *et al.* (2008) report a speculative simulation for Pontbren using the extreme rainfall that generated the Carlisle flood of January 2005. The rainfall total over 2 days was 140 mm, with an estimated return period of 180 years.

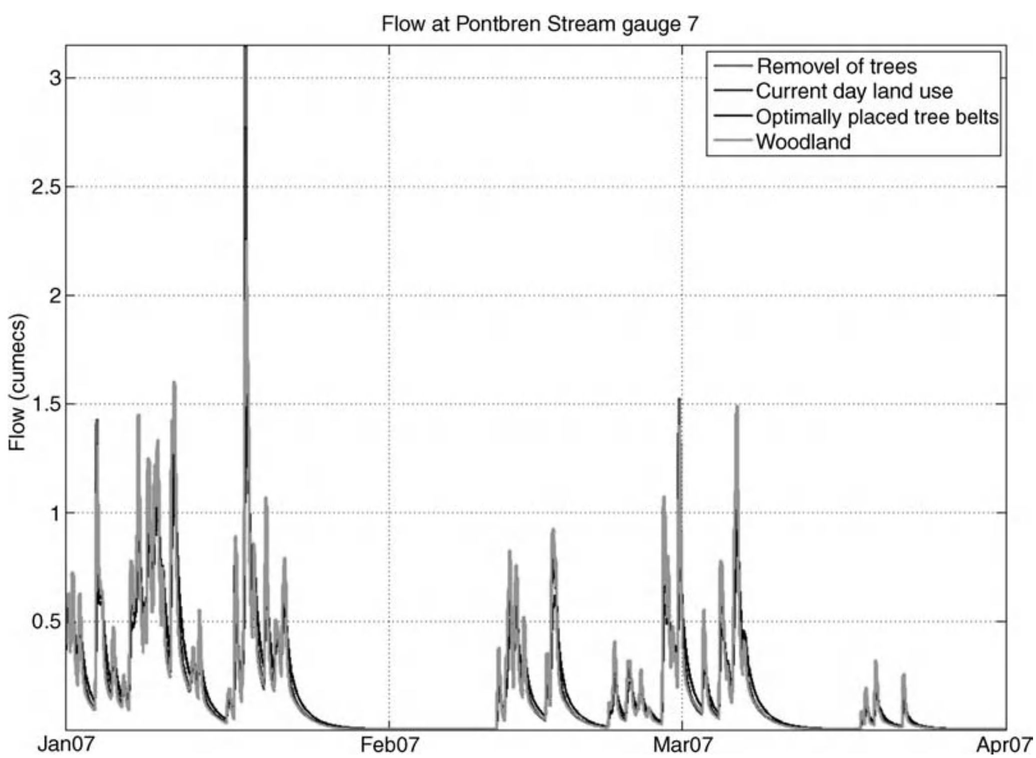

Fig. 3.11 Catchment-scale response for a 4-km^2 Pontbren subcatchment for current land use and a set of scenarios: 1990s intensification, further addition of tree shelter belts, and full aforestation. (see the colour version of this figure in Colour Plate section.).

There was quite large uncertainty in these results, due to parameter uncertainty; however, median results showed that:

- Removing trees planted within the last decade causes a 3–7% increase in flow peaks from the baseline condition.
- Adding tree shelter belts across the lower parts of all grazed grassland sites causes a 2–11% decrease in flow peaks from the baseline condition.
- Afforestation of the whole catchment causes between a 10–54% decrease in flow peaks from the baseline condition.
- There was no apparent uncertainty in time-to-peak results, due to the 15-minute resolution of the model. The tree removal reduced time-to-peak by 15 minutes, woodland cover increased time-to-peak by 30 minutes, while the shelter belts had no effect.

Model Regionalization

Introduction

Pontbren is one of only a few catchments in the UK that are intensely monitored. Clearly, for more general application, methods are required to quantify response for other areas, land management interventions and scales. This raises issues of the relative lack of detailed supporting data for other environments, the potential role of physics-based methods in data-sparse areas, and the challenge of regionalization. These issues are being addressed in ongoing research under the FRMRC2 and FREE programmes. Current progress is reported below.

Physics-based modelling in data-sparse areas: the representation of upland peat management

As we look to extending the upscaling modelling approach to catchments other than Pontbren, one of the greatest challenges is how to expand the existing metamodelling library to account for the range of land use, land management and soil types that will be encountered in new catchments. For the Pontbren catchment, detailed physics-based models were developed and conditioned using the data from an intensive monitoring programme. However, it is very unlikely that such extensive datasets will be available for the development of all of the possible metamodels required for the analysis of new catchments. The critical question becomes what is the role of physics-based models in data-sparse areas?

Computationally intensive physics-based models provide the capability to simulate explicitly the effects of local changes, for example due to localized tree shelter belts located within a field or hillslope unit. And, despite significant data scarcity, these models may still be the best possible way to upscale local changes to catchment scale, given that our understanding of the impacts of land use and land management changes is largely restricted to the very small scale (such as changes in water retention properties, interception and runoff processes). While continuous hydrological measurements may not be available, useful information about small-scale hydrological processes can still be obtained from the literature. The extent to which uncertainty can be constrained by such data is a key research question. We also note that physics-based models have the power to support the development of improved conceptual understanding of runoff processes and the dominant physical controls, and can thereby provide qualitative insights that may be of value when considering effects of land management change.

A methodology is therefore proposed for the development of detailed physics-based models for data-sparse land use and land management types. The key hydrological processes should be identified from the literature and included in a physics-based model that has a level of complexity that is appropriate relative to the level of detailed information available on the system hydrological processes. To avoid over-parameterization, minor processes could be excluded or treated in a simplified manner. Examining the model behaviour should also then provide some further insight into the relative significance of the model parameters and also inform the metamodelling process.

This approach has been trialled for the development of a detailed physical model of upland peat management, for specific application to the Hodder catchment in northeast England. In the UK, approximately 1.5 million ha of the country's 2.9 million ha of peat has been drained (Milne and Brown 1997). Drainage of peats is typically via a series of open ditches, alternatively running parallel to the slope, in a herring-bone pattern or randomly positioned (Robinson 1990). This particular type of drainage is often referred to as 'gripping' (Stewart 1991). Peats are drained with the aim of improving vegetation and therefore the production of livestock and game (Stewart 1983). The rationale is that drainage will remove excess runoff and lower the water table. Recent work has evaluated the effects of grip drainage and grip-blocking, using surrogate data from research sites elsewhere (Wallage 2007). Current work is developing metamodels for catchment-scale application.

Regionalization approaches and preliminary results

As discussed above, an important research objective is to develop a regionalization scheme that may be applied throughout the UK to predict flows in ungauged catchments and to explore impacts of local land management changes to catchment properties. As a first step towards this, the response index approach to regionalization is adopted (Bardossy 2007; Yadav *et al.* 2007), in which indices are used to condition prior model parameter uncertainty into posterior distributions so that probabilistic predictions of land management impacts can be made. While the methodology allows all available runoff response indices to be introduced, we use the Base Flow Index, derived from the HOST classification of UK soils (BFI_{HOST}) with the hypothesis that this on its own is usefully informative (Lamb and Kay 2004; Lee *et al.* 2006; Young 2006; Yadav *et al.* 2007). The strategy is to use BFI_{HOST} to constrain model parameters (Bulygina *et al.* 2009). This then provides a basis for consideration of modifications to HOST to allow for changing land management practices.

Methodology

The parameter space restriction methodology is as follows. Suppose a catchment is discretized into a large number of runoff-generating elements, and catchment response is viewed as the integration of all the individual elemental responses. Potentially, the catchment model needs a separate set of parameters for each element. Here, it is assumed that all elements with the same BFI_{HOST} have the same set of parameter values. Therefore, the number of different parameter sets needed for runoff generation modelling is far less than the number of elements and cannot exceed 29 – the number of soil types in the HOST classification.

For each model element, a hydrological model can be run, and then the BFI can be estimated from the simulated runoff (Gustard *et al.* 1992). The simulated BFI is compared to the expected BFI_{HOST} value for the associated soil class, accounting for the standard deviation of BFI_{HOST} due to natural variablity within a soil class as defined by Boorman *et al.* (1995). This comparison is used to identify behavioural and non-behavioural parameter sets, and the associated likelihood.

Modelling aspects of land use change involves identifying suitable changes to parameter distributions for the affected elements. Two examples are discussed here – afforestation and intensive grazing. There is also evidence that base flow proportion increases under forest (Wheater *et al.* 2008), but currently only limited quantitative information about this change is available. Therefore, afforestation is assumed to lead to higher BFI, while keeping the same HOST soil type. Changes in interception losses associated with afforestation may be estimated using a standard model using parameter distributions from the literature (David *et al.* 2005; Jewitt 2005). The second land management change considered is increasing stocking density, leading to soil structural degradation. Following the approach of Hollis (Packman *et al.* 2004), degraded soil is assigned an appropriate analogue HOST class to represent the change.

Application to Pontbren

The time-series data used for this study are 15-minute resolution rainfall, daily MetOffice Rainfall and Evaporation Calculation System (MORECS) evapotranspiration data, and 15-minute resolution streamflow data from six gauges (gauges 2, 5, 6, 7, 9 and 10: see Fig. 3.1). Gauge 10 data are used for assessing low flow ($< 1.5\,m^3/s$) performance only because the rating curve for higher flows is poorly defined. The contributing areas at each of the six gauges are given in Table 3.2. The period used for the modelling demonstration is 1 January 2007 to 1 April 2007, for which the best-quality and most complete data exist.

The catchment is discretized into $100 \times 100\,m$ runoff-generating elements. A semi-distributed modelling toolkit (Wagener *et al.* 2004; Orellana *et al.* 2008) is used that requires specification of a conceptual runoff-generating model and a routing model for each element. A probability distributed soil moisture model together with two parallel linear routing stores (Fig. 3.12) is selected, as this is perceived to be widely applicable in the UK (Lamb and Kay 2004; Calver *et al.* 2005; Lee *et al.* 2006).

Table 3.2 Contributed areas for the considered gauges

Gauge number	2	5	6	7	9	10
Contributing area, km^2	1.3	2.4	3.2	5.8	4.1	12.5

There is a lake upstream of gauge 8. Its response is simulated using the weir equation $Q = kH^m$, where H is the water level in the lake above the outlet's lowest point, and the coefficients k and m are related to the outlet geometry (Montes 1998), which, following channel measurements (McIntyre and Marshall 2008), is assumed to be parabolic.

Each runoff-producing element is prescribed a parameter set according to BFI-based regionalization described above. Element runoff is routed down the stream network using a constant celerity approach, i.e. the water moves with constant velocity (Beven 1979). Several samples of peak flow arrival time observed at gauge 6 are used to estimate the best of the sampled celerity values.

Figure 3.13 shows discharge predictions for January 2007, which is representative of the 3-month evaluation period. Here, the dark-grey areas represent the 90th percentile on the discharge prediction; and the black dots are observed data points. The light-grey area represents the 90th percentile of prior uncertainty, when there is no distinction due to soil type and land use, and parameters are assigned uniformly across the catchment. The dashed lines show the range of flows within which the streamflow gauge was calibrated and considered accurate (McIntyre and Marshall 2008), so that the data points lying in the range could be considered as being more reliable than the points lying outside. Note that gauge 10 rating curve is estimated using flows up to $1.5\,m^3/s$ only. The Nash–Sutcliffe statistics for expected values are summarized in Table 3.3. The

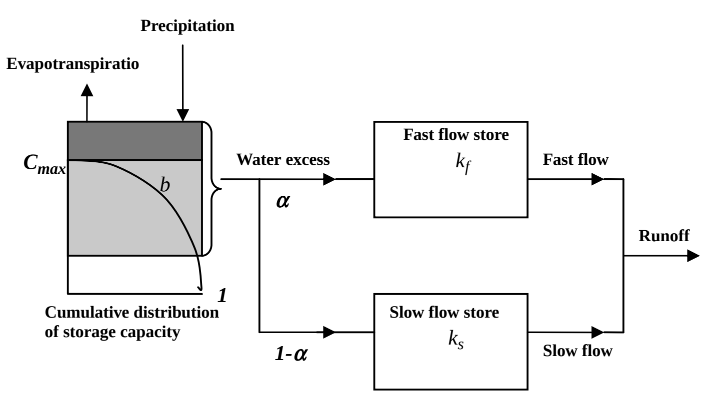

Fig. 3.12 Rainfall-runoff conceptual model and associated parameters.

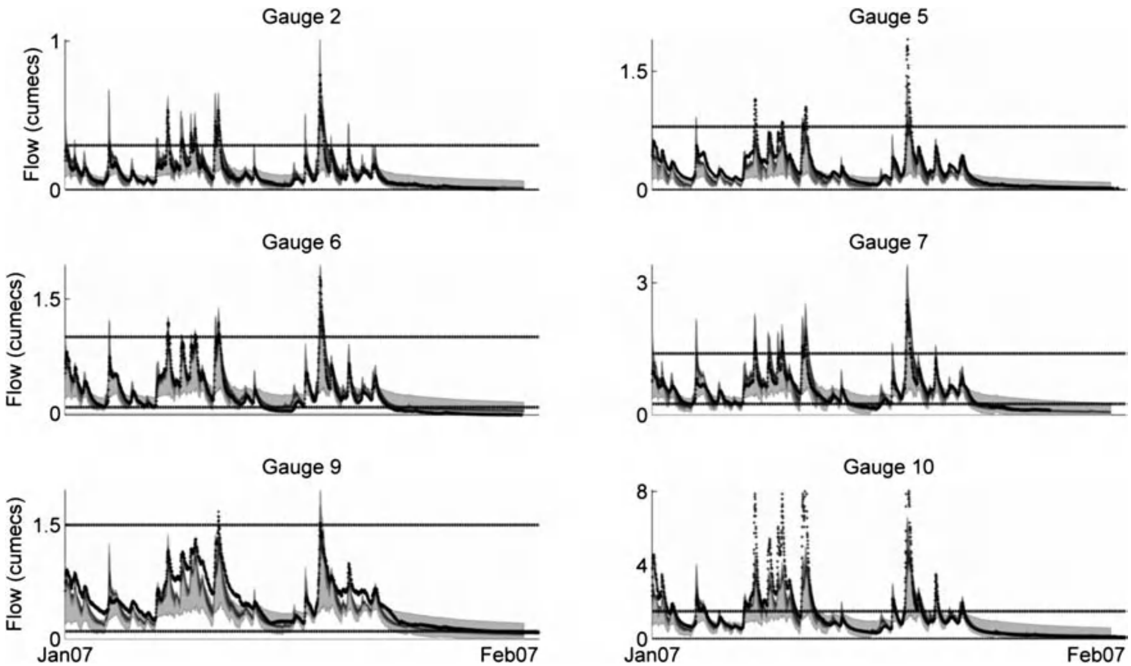

Fig. 3.13 Prediction uncertainty bounds for January 2007.

performances achieved together with Figure 3.13 support the view that BFI_{HOST} is an effective response index.

Figure 3.14 shows the predicted impacts of these land management changes on runoff at gauge 10. The median relative differences associated with the scenarios are given in Table 3.4: changes in total runoff within the period 1 January 2007 to 31 March 2007, and changes in the highest observed peak flow during the period (18 January). The afforestation delayed the highest peak arrival by 15 minutes (one simulation time step), and the soil degradation scenario did not show any difference in peak flow arrival time.

Table 3.3 Nash–Sutcliffe efficiency coefficients (1 January 2007 to 31 March 2007)

Gauge 2	Gauge 5	Gauge 6	Gauge 7	Gauge 9	Gauge 10
0.84	0.70	0.85	0.78	0.80	0.65

This study illustrated a simple method of conditioning hydrological model parameters on prior information in order to simulate runoff under current conditions and future land management scenarios. The prior information about current conditions, in this case, came almost entirely from the BFI_{HOST} index from a national database of soil types. The conditioned model was shown to simulate observed flows to an impressive level of accuracy. The approach described in this paper has allowed impacts of land use management interventions to be predicted. Two scenarios were investigated: (i) returning the catchment to a pristine woodland landscape through afforestation; and (ii) further degradation of the soil associated with overgrazing. Median values show significant impacts: for example, the changes in the largest observed flood peak at the catchment outlet were a 12% reduction with afforestation and an 8% increase due to overgrazing.

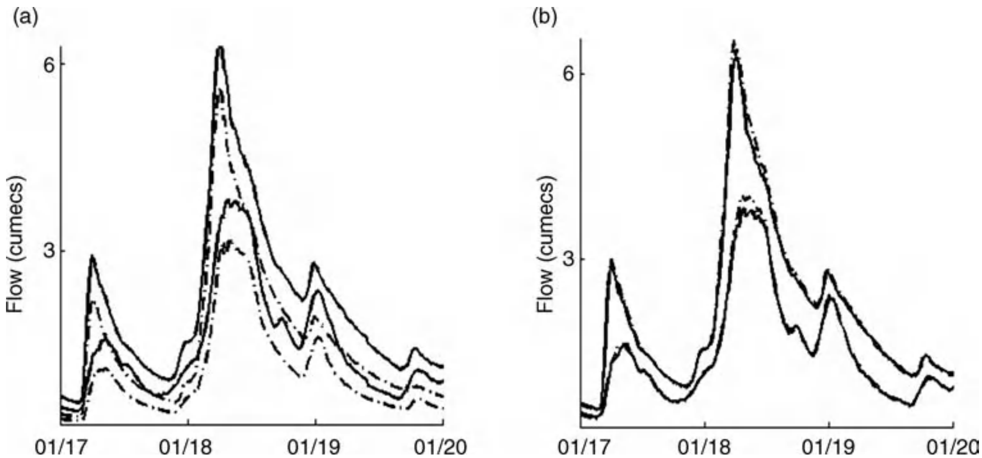

Fig. 3.14 Prediction uncertainty bounds at gauge 10 due to the 18 January 2007 rainfall event: (a) afforestation, (b) soil degradation.

Summary and Conclusions

The results of an intensive experimental programme have been reported from the Pontbren multiscale experiments. Detailed characterization of soils and runoff processes has provided data to constrain models developed to represent both local-scale and catchment-scale responses. A complex story emerges, with important roles for drain and overland flow as the primary runoff-producing mechanisms from improved pasture. Typically the low-permeability B horizon restricts downward movement of water within the profile and overland flow occurs due to A-horizon saturation. However, important effects of interannual variability were observed, with a distinctly different response following the dry 2006 summer, and as the observation period has lengthened, seasonal variability has been discerned. Unimproved grassland provides a significantly more damped runoff response.

Based on the Pontbren data, a multiscale modelling procedure has been developed. At the scale of a field or hillslope, highly detailed physics-based models provide the basis to predict effects of small-scale management interventions, such as tree shelter belts. Simple metamodels appear to capture well the response of the detailed physics-based models, and provide the basis of representation of individual fields within a distributed

Table 3.4 Median value of relative reductions in total flow and peak flow (in percent relative to current-day land use)

	Gauge 2	Gauge 5	Gauge 6	Gauge 7	Gauge 9	Gauge 10
Afforestation						
% **reduction in total runoff**	24					
% **reduction in peak flow**	14	13	12	14	15	12
Soil degradation						
% **reduction in total runoff**	0					
% **reduction in peak flow**	−9	−9	−9	−9	−4	−8

catchment-scale model. Hence simulations have been used to quantify effects of a range of land management interventions (with uncertainty) for both frequent and rare flood events.

Pontbren results are site-specific, and much work is required to provide a more general methodology for national application. First steps towards this are presented. Detailed physics-based models of the peat provide insight into the effects of local management interventions (such as grips and grip-blocking), but there is a lack of local data to constrain parameterizations. Work is ongoing to assimilate data from more intensively monitored sites. The resulting uncertainty has yet to be quantified, but this work will address an important research question, namely the role of physics-based models in data-scarce situations.

Regionalization of results must depend on nationally available data, and the use of the HOST soil classification has been investigated. Use of a Base Flow Index based on HOST has proved a powerful tool to constrain model parameterizations, and potentially provides a connection to the work of O'Connell *et al*. (2004). If, for example, soil degradation can be represented as a change in HOST class, a direct connection can be made to the parameterization of catchment-scale models. However, more work is needed to explore the role of physics-based models in providing information to support the definition of appropriate changes to the regionalized parameters.

Finally, we note that current research links the above with work at the University of Newcastle upon Tyne into information-tracking algorithms (see Chapter 0), and this then provides a powerful methodology not only to predict catchment-scale changes, but also to analyse the local sensitivity to management interventions.

Acknowledgements

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4 Managed Realignment: A Coastal Flood Management Strategy

IAN TOWNEND, COLIN SCOTT AND MARK DIXON

Introduction

The removal of existing flood defences has been variously referred to as managed retreat, managed realignment and habitat creation or restoration, depending on the underlying objectives of the particular scheme. For the purposes of this review, managed realignment is the form of coastal adaptation that removes a part, or all, of a sea wall in order to allow some additional land area to be subject to tidal action. This may, or may not, require the provision of modified defences, or defences set back on a new line, to protect local assets.

The primary motives for carrying out managed realignment are to adapt to sea level rise, to enhance coastal protection levels, to create a more cost-effective defence alignment and/or to create new coastal habitats. The habitat creation objectives are, in turn, driven by the need to compensate for losses following coastal developments or to restore and conserve habitats that have been, and are, subject to more general deterioration. This deterioration of coastal habitat is often attributable to 'coastal squeeze', and is a process that has been exacerbated by past land reclamations in many instances. On unprotected coasts, as sea level rises, the shore is able to move landwards, giving rise to what is called marine transgression. However, around many coasts and estuaries, the existing defences fix the interface between land and sea. By preventing the process of transgression from taking place, it is often the case that the more seaward part of the shore profile continues to move landward but, because the upper part of the profile is held by the sea defences, the intertidal, in front of the defences, gets narrower (Taylor *et al*. 2004).

Conservation and restoration of deteriorating coastal habitats, alongside project-specific compensation measures, have been the dominant motives for managed realignment. Across northern Europe some 76 realignment schemes have been completed over the last 20 years, and in this time there have been a further 18 Regulated Tidal Exchange (RTE) projects, where the tidal flows are managed through control structures such as sluices or weirs rather than full openings of the sea wall. While many of these 94 schemes had more than one objective, their primary motives have been conservation (33 schemes), compensation (32 schemes), flood defence improvement or cost reduction (16 schemes) or the development of a more natural shoreline (four schemes) with the primary driver for the remaining nine schemes being unknown (Rupp-Armstrong *et al*. 2008, Rupp-Armstrong and Nicholls 2010). In the USA almost all the managed realignments (alternatively referred to as 'restoring tidal inundation' or 'dike/levee breaching') have been undertaken for conservation or compensation reasons (Rupp 2009). Outwith these primary aims for managed realignment, there are, as we shall discuss later, many socioeconomic benefits of undertaking habitat restoration work that go beyond these key aims and, equally, many such socioeconomic impacts from the ongoing habitat deterioration. As

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one example, Andrews *et al*. (2000) concluded that 90% of the intertidal area has been lost to land claim on the Humber Estuary and, consequently, that 99% of the storage potential for carbon in the estuary has also been lost.

The need for managed realignment schemes is often driven by, or at least supported by, a range of national and international legislative and policy drivers. In Europe, the EU Habitats Directive requires compensation to be provided, ideally as close as possible to the area of loss, where developments or strategies are deemed to affect the integrity of internationally designated Natura 2000 sites and Ramsar wetlands. This can include compensation for coastal defence management plans where these contribute to ongoing deterioration through coastal squeeze. Another international policy driver for the offsetting historic and ongoing coastal habitat losses is Biodiversity Action Planning (BAP), for which objectives for the restoration of saltmarsh, mudflat and saline lagoon habitats have been identified in response to the 1992 Rio Convention on Biological Diversity (CBD). For example, the UK government has BAP targets for saltmarsh and mudflat creation that add up to 600 ha per year combined (UKBAP 2006), with an additional target to create a further 3600 ha by 2015 to offset historic losses.

In the UK there are also targets that seek to restore coastal habitats within nationally designated Sites of Special Scientific Interest (SSSIs) to a favourable conservation status, and one current target is to restore 95% of SSSIs to this status by 2010. Also, the creation of new habitats will contribute to improving the status of estuaries under the European Water Framework Directive (WFD) through the enlargement of the intertidal zone per se, and the subsequent increased abundance of angiosperms (i.e. seed-bearing saltmarsh) and the likely improvement of conditions for fish populations.

In the USA comparable legal drivers exist including the Clean Water Act (1968), which requires compensatory mitigation for harmful discharges into wetlands, and the Coastal Wetlands Planning, Protection and Restoration Act (1990), which drives the protection of existing wetlands and restoration of lost habitats (Rupp 2009).

These policy drivers and targets for coastal habitat creation provide clear national and international strategic rationale(s) for managed realignment; perhaps the most quantifiable, and hence probably clearest, rationale for continuing such projects is demonstrated in the UK by the shortfall that exists between the amount of habitat created so far and the habitat extent targets that have been set. Against the targets set out above, the realignment and RTE projects completed by 2009 have created 1330 ha of new habitat. A third of this area (450 ha) was for project-specific compensation and therefore does not represent restoration, while the remaining two-thirds (880 ha) was for nature conservation, biodiversity enhancement, coastal squeeze compensation as well as flood defence enhancement (Rupp-Armstrong *et al.* 2008) and these collectively contribute to a restoration target. To illustrate the contribution that has been, and can be made, to habitat creation targets specifically by managed realignments, these 1330 ha were created by 44 projects of which 33 were realignments delivering 94% of the habitat area while the remaining 11 projects were smaller-scale RTEs contributing 6%.

Although a national and international strategic rationale for implementing these schemes exists, the dichotomy faced by coastal managers who seek to implement them is one of how to balance the need to protect assets that are currently behind defences, be they farmland, property or infrastructure, with the need to give the shoreline sufficient space to respond to the changing conditions and so maintain a healthy suite of habitats within the transition from sea to land. This conflict between the costs of land protection and the value of the land has a long history, with the debate shifting according to prevailing social and economic priorities. Soft coastlines have always changed but the idea of seeking to fix them and/or claim new land to create new agricultural land relatively cheaply was clearly prevalent from the 14th to 20th centuries. In the 20th century, however, priorities shifted several times. The 1909–1912

Royal Commission on Flood Defences, for instance, deemed that state aid to restore land that had been subject to unmanaged flooding was too expensive in most areas. The aftermath of World War I saw priorities change when the Bledisloe Commission decided that the UK needed to be self-sufficient in food in the event of further conflict, and provided state funding for draining wetlands and keeping saltwater out. This policy was reversed after the 1953 floods when the Waverley Commission recommended that public funding should not be used for agricultural land on large parts of the East Anglian coast because the economic returns were too small. This policy was never put into practice though and large sums of money continued to be spent for the benefit of small numbers of coastal farmers. Today, population increases (nationally and globally) mean that our attention is again focused on whether we are going to be able to provide sufficient food into the future. But, that cannot mean that we must protect all extant coastal farmland, especially when the food produced on relict saline floodplains protected by sea defences is among the most expensive in the world, and with certain exceptions (e.g. Wash and Somerset Levels) is on a relatively small area of land. Instead it must mean that there is a prioritization of funding and land use in the context of a national strategy.

To inform the contemporary debate on land loss and managed realignment there are values that can be ascribed to both the losses and the gains associated with the changing habitats, but they need to be weighed alongside a number of intangible benefits (existence value, recreational value, etc.). So, for example, one may need to consider the value of some high-grade farmland, both now and in the future, against the value of some intertidal mud and the services that it provides (areas of intertidal mud within estuaries are some of the most biologically productive areas on the planet and are of great importance to the marine food chain). Such trade-offs can be complex and have often led to difficulties when coastal managers have sought to promote and consult upon managed realignment schemes. For instance, in 2003, a proposal to carry out managed realignment at Weymarks in the outer Blackwater Estuary met with robust opposition from the local community and the planning authority due to its perceived 'significant effect on the character and appearance of the countryside' in an area of coastal farmland (Maldon District Council 2003). Other concerns were also expressed relating to flood risk, the adverse effect on the local economy, the damage to local wildlife interest (on arable fields), the loss of access to a small shingle beach and the effects on features of archaeological interest. Ultimately, this opposition led to this site being abandoned in favour of a 115-ha realignment on the north bank of nearby Wallasea Island (ComCoast 2007, Dixon *et al*. 2008).

Problems were also encountered with the 600-ha Kruibeke flood alleviation scheme, on the banks of the Scheldt Estuary near Antwerp in Belgium (still under development in 2009). While not a managed realignment site, it represents a large-scale (600 ha) area that has some RTE areas with new counterwalls that will be designed such that the outer walls can safely overtop on surge tides to alleviate flood pressures elsewhere. Although the scheme had first been mooted, and known publicly, in the late 1970s there was an absence of consultation with the public or the media, and the information vacuum fuelled political opposition led by the local mayor (ComCoast 2007). This was addressed through the instigation of a detailed communication exercise led by a dedicated communications team and included the provision of a drop-in visitor centre on the construction site. By contrast, and as just one example of how lessons have been learned from the past, the implementation of a 370-ha flood alleviation and managed realignment scheme at Alkborough on the Humber Estuary was helped greatly by having public consultations/involvement including the creation of a liaison group with local representatives that met regularly throughout the duration of the project.

In addition to the lessons that have been learned about how to engage with the community, our understanding about the socioeconomic benefits that accrue from newly created habitats and how to value them in monetary terms has been significantly enhanced in recent years. This has come from a combination of research into the performance of completed schemes and new understandings about the valuation of ecosystems. In the past, for instance, there was confusion about which elements within an ecosystem to value and how to cost for these, but this impasse has been unlocked by recent assessment and guidance documents, such as the UN Millennium Ecosystem Assessment (MEW) reports and the UK government's guide to valuing ecosystem services (Defra 2007). The guidance sets out a method to describe what an ecosystem does for the 'human community' by recognizing that the biological diversity contributes to 'wellbeing' but framed in economic terminology (Watts and Kremezi 2008). Therefore, whereas in the past the economic value of managed realignments was dictated by the flood defence costs, it is now possible to quantify the wider economic gains from habitat creation. Case example valuations have been made using the research findings from completed realignments such as Abbotts Hall on the Blackwater Estuary (Salcott Creek) and both Paull Holme Strays and Alkborough on the Humber Estuary (Kremezi 2007) and for the recently consented (July 2009) Wallasea Island Wild Coast Project on the Roach Estuary (Eftec 2008). These studies considered the socioeconomic benefits from fisheries, carbon storage, nutrient processing, pollutant sequestration and job creation to indicate that managed realignment schemes can be justified by the economic gains that they provide alongside their flood and coastal defence benefits.

Whatever the objectives when developing a scheme, what has become very clear is that these need to be identified at the outset. This then greatly facilitates the processes of site selection and designing suitable site layouts and, crucially, the consultation process. As with any development, there are a number of requirements that need to be met to promote the scheme through the planning system and so obtain the necessary consents and licences. Once obtained, construction can commence and this involves working in or next to the marine environment, which brings its own challenges, notably where this entails intertidal working. Finally, having completed the works, there is invariably a need to monitor how the site performs and to evaluate how well the scheme objectives have been met. As part of the monitoring process it is important that the findings are communicated widely. The process of collating and disseminating the lessons learned also helps to improve the process of implementing future schemes. In this chapter, the main steps involved in promoting a managed realignment scheme and taking it to completion through these stages are reviewed under the following subheadings:

- Setting scheme objectives
- Selecting a site for realignment
- Designing the scheme (including identifying design constraints, characterizing the existing site conditions and developing the design layout)
- Obtaining planning approval and consents
- Undertaking the construction work
- Monitoring and evaluating scheme performance

It is hoped that this stepwise review of the relevant issues at each key step in the lifetime of such projects will, alongside the supporting literature, provide a useful and practical guide that will help coastal managers to implement such schemes more effectively, and perhaps, more importantly, to improve the quality, quantity and, where possible, the size of such schemes in order to maximize the benefits for the environment, the economy and society.

Setting Scheme Objectives

Of paramount importance in habitat creation or restoration schemes is the setting of clear aims and objectives that define what the scheme is principally trying to achieve. The schemes that have been successfully promoted and consulted upon have usually benefited from having this clarity of purpose and, ideally, from a reliance on simple, consistent and non-technical messages. These primary objectives might encompass the basic rationale and more specific delivery needs. Previous schemes have been undertaken for various reasons of which the main ones, as discussed above, are usually restoration/replacement of coastal wetland and the implementation of more sustainable coastal flood defences.

It is vital that objectives are clearly identified at the outset, not only because they can influence the design and planning process but also, and most importantly, clarity is essential in any consultation or public engagement to promote the scheme. Numerous reviews of previous schemes have also identified the fact that failure to be clear about what the scheme is trying to achieve, and agreement on this among the stakeholders, is the most common reason for schemes not meeting expectations. To this end it is important to consult not only with the scheme promoters but also with the local community, relevant conservation bodies and consenting authorities from the outset. Further guidance on the best practices for consultations is available separately through projects such as ComCoast (2007).

In addition to the main objectives of the scheme, which need to be clearly laid out, there are other secondary benefits that can accrue. Indeed, wherever possible, secondary benefits and multiple uses should be identified and brought into the design. Recognizing these other benefits and highlighting them can help to broaden the appeal of a managed realignment initiative by ensuring that the local community can be brought on board and can feel involved in the process. New lessons are being learned all the time about these other benefits as a direct result of practical experience and scientific research (Andrews *et al*. 2006; Shepherd *et al*. 2007; Shih and Nicholls 2007; ComCoast 2007; Watts and Kremezi 2008; Dixon *et al*. 2008; Rupp 2009). A selection of the different project motives with corresponding primary or secondary objectives is shown in Table 4.1.

In each case more detailed subsidiary targets will be relevant under these headline objectives, and these could encompass such things as maintaining habitat connectivity; the delivery of particular species and habitats; the provision of a continued level of flood protection to adjacent land areas; the avoidance of impacts from sea wall failure, as well as other specific hydrological, geochemical, morphological and biological requirements

Table 4.1 Main reasons for, and benefits of, managed realignment schemes

Coastal defence and conservation	Environmental quality	Socioeconomic
Enhanced flood protection in the short term (reduced flood risk)	Improving water quality through nutrient processing (e.g. nitrate removal or silica cycling)	Commercial and recreational fisheries by providing foraging and nursery area for fish species such as bass, herring and flounder
Enhanced flood protection in the long term (by allowing room for adjustment with sea level rise)	Carbon sequestration (as new habitat develops and accretes it absorbs carbon)	Shellfish culture for species such as oysters, cockles and mussels
Reduced costs for flood protection	Other air quality benefits from 'wet' and 'dry' absorption of particulates and some ozone precursors	Green tourism (e.g. bird watching, footpaths, cycle paths, bridleways)
Compensation for habitat losses following individual developments or coastal strategies	Sequestration of nutrients and other pollutants such as heavy metals	Wildfowling
Nature conservation and biodiversity enhancement/restoration	Improvements to the morphology and landscape (e.g. by reinstating natural habitat transitions) and regeneration benefits where new wetlands are created in urban areas	Recreational boat moorings and water sports
		Sea-salt or marsh samphire harvesting
		Livestock grazing (e.g. for 'saltmarsh lamb')
		Archaeological conservation
	Location for the 'beneficial use' of dredge arisings and other suitable materials	

Selecting a Site for Realignment

When selecting locations for realignment there are a number of site characteristics that are relevant, of which the existing land use and the land elevation in relation to the tidal frame are probably of greatest importance. The existing land use is clearly critical and will influence the economic rationale for the work and the feasibility of undertaking the projects, with aspects such as the presence of infrastructure, the proximity of road or rail lines, and the alignment of gas and electric pipelines or pylons all being relevant. Land elevation is also important both to engineering feasibility and to potential nature conservation outcomes. Other considerations include minimizing the engineering costs; reducing the length, and maintenance cost, of the new sea wall; the potential to impact on the wider estuary or coast; or the effect on the stability and protection afforded by sea walls in the vicinity (Dixon *et al.* 2008).

These aspects usually form the core selection criteria but numerous others may also be relevant depending on the scheme objectives, as indicated in Table 4.1. The site selection process also varies, most obviously, in terms of the spatial coverage and level of detail applied; for instance, they range from:

- **High-level strategic audits of the national habitat creation resource:** An example of this includes the Royal Society for the Protection of Birds' (RSPB's) review of intertidal habitat creation sites across mainland Britain, which used criteria such as the absence of infrastructure and the need to avoid increasing the length of flood defence (Pilcher *et al.* 2002). This study identified an area of around 33,000 ha that was potentially suitable for realignment.
- **Individual strategies covering a single estuary or coastal management area:** Such studies are designed to identify a set of sites that will contribute to the strategic management of coasts and estuaries while also providing compensation sites to offset past habitat losses and the effects of coastal squeeze. They include Shoreline Management Plans (SMPs); Coastal Habitat Management Plans (CHaMPs) such as those produced for the Thames (ABPmer 2008) and the Solent (Bray and Cottle 2003; Hodge and Johnson 2007) as well as Flood Management Strategies such as the Essex Estuaries Strategy (Ahlhorn and Meyerdirk 2007) or the Solent Dynamic Coast Project (Cope *et al.* 2007).
- **Project-specific compensation site reviews:** These are studies designed to identify a single compensation site that will offset habitat losses from a specific coastal development, in which case the search is centred on the location where the losses take place. A relatively detailed example of this includes the UK government's search for compensatory sites that would offset loss of intertidal habitats from port developments at Lappel Bank (Medway Estuary) and Fagbury Flats (Orwell Estuary). This search extended from north Kent to southern Suffolk and ended with the selection of the 115-ha Wallasea North Bank realignment site that was breached in 2006 (ABPmer 2003, 2004a).

For simplicity, the process is most easily viewed in two stages. The first stage is a 'Screening' process at which searches of varying detail (national, regional, coastal cell, estuary) are undertaken usually making use of geographical information system (GIS) mapping techniques. The second stage is then a 'Scoping' exercise during which the feasibility of an individual site, or selection of sites, is considered through consultation and site investigation, as set out in the next section.

Most screening studies begin with a floodplain map and use a range of criteria to select the most suitable sites (e.g. by avoiding built-up areas, roads or railways; identifying areas with elevations suitable for intertidal habitat creation, and considering land use and land ownership issues). To understand the sorts of issues that are relevant, Table 4.2 shows the criteria that were applied in a range of studies. In most instances the criteria used are broadly similar but there is often considerable variability in the emphasis that was put on physical and/or anthropogenic factors as well as in the stage in the hierarchical process at which certain criteria have been used. Such flexibility is to be expected given the variability of objectives, and the list in Table 4.1 can be seen as

Table 4.2 Site selection criteria used for screening and scoping exercises (different criteria weightings were applied in each case)

Screening (SCR) and/or Scoping (SCO)	SCR & SCO	SCR	SCO	SCR	SCR & SCO	SCR & SCO	SCR & SCO	SCR & SCO
Location/extent	RSPB Review of British resource	Solent CHaMP	Solent	Thames CHaMP	CEFAS Tool	Lappel Bank/ Wallasea	Essex Estuaries Strategy	Solent Dynamic Coast Project
Criteria/reference	Pilcher *et al.* (2002)	Bray and Cottle (2003)	Hodge and Johnson (2007)	ABPmer (2008)	Parker *et al.* (2004)	ABPmer (2003, 2004a)	Ahlhorn and Meyerdirk (2007)	Cope *et al.* (2007)
Suitable elevation (based on floodplain/ 5-m contour)	x	x		x		x	x	
Suitable elevation (based on tidal levels)	x			x	x	x		x
Suitable slope					x			
Proximity to existing habitats					x			
Proximity to lost habitats (compensation)						x		
Presence of contaminated land/landfill			x		x			x
Presence of areas within 2 km of landfill site								
Presence of built-up/urban areas	x	x		x		x		x
Presence of main roads	x		x	x				
Presence of railway lines	x		x	x				
Presence of woods				x				
Presence of designated sites			x	x		x		
Length of new counterwall relative to existing wall	x					x		
Water salinity, freshwater flows and water quality					x			
Biological/propagule supply					x			
Habitat location (either within estuary or along coast)	x				x			
Exposure and connectivity (i.e. homogeneity/drainage)					x			
Bed stability and soil type					x			
Standard of flood defence and need for new/secondary defences	x		x			x	x	
Number of years embanked			x					
Minimum area for site and/or amount of specific habitat delivered	x					x		x

Morphological functioning, sustainability and effects on hydrodynamics			x	x	
Economic viability, engineering feasibility and costs	x		x	x	x
Agricultural land use quality and land ownership issues (e.g. number of owners)		x	x	x	x
Potential for environmental improvement				x	
Potential effects on terrestrial/freshwater habitat			x		
Heritage/archaeology interests					x
Presence of abstraction licences					x
Rights of Way					x
Potential political acceptance		x			

Centre for Environment, Fisheries and Aquaculture Science (CEFAS), CHaMP, Coastal Habitat Management Plan.

a menu from which future studies can select the criteria that most effectively address the project needs.

The extent to which screening is required varies between projects in response to the availability of the appropriate strategic information. Where a coastal or estuarine strategy is already in place then much of the screening may well have been done and only the scoping phase needs to be carried out. Thus the process of searching for individual sites can be greatly accelerated, and the number of iterations reduced. This is because the strategy should provide the rationale for site selection, so that only the more detailed work to identify site-specific issues and constraints needs to be undertaken. Equally, the availability of a strategy can itself facilitate the consultation process because it should set out the reasons why a particular location has been selected.

Designing the Scheme

There are usually a number of options as to how a particular site is developed, although the most common is to simply breach the existing wall and create a new area of mudflat and saltmarsh to the rear. Alternatives include tidal creeks, breached sea wall or complete removal of the sea wall, managed breaches with sluices to control flow, saline/brackish lagoons, and the provision of flats and islands. The choice will depend on which option, or combination of options, best meets the objectives for the site and the constraints that are invariably posed by the specific setting of the site.

Typically the design process will encompass the following steps:

1 confirmation of design objectives;
2 identification of design constraints;
3 selection of site (if not predetermined);
4 characterization of the site;
5 determination of target habitats and species (if required);
6 field investigations;
7 develop site layout and design any features to be constructed;
8 sensitivity and risk assessment;
9 construction programme, costings and tender documents; and
10 design and specification of monitoring programme.

These various aspects of the design process are presented in the form of a flow diagram in Figure 4.1.

Identifying design constraints

A number of physical and ecological factors will limit or constrain what can be achieved at a given location. The relevant design constraints must therefore be identified at the outset of the design process. For a managed realignment the constraints might typically include various physical and habitat constraints:

Physical constraints

- maximum area, length and width of the site;
- existing level of site in tidal frame;
- potential to remove sea wall;
- surrounding bathymetry, extent of adjacent intertidal and distance to main channel;
- underlying geology and lithology;
- availability of suitable surficial sediments;
- freshwater input.

Habitat constraints

- habitat value of existing site and adjacent sites;
- target species and habitats (if a compensation or mitigation scheme);
- exposure to waves (storms and ship wake);
- water quality and sediment quality;
- levels of disturbance.

Characterizing the existing site conditions

The success of the created habitat will be dependent on the existing physical, chemical and biological characteristics of the site and adjacent environment. Much of this information will need to be obtained through field surveys, literature review or the analysis of existing data sources. Investigations will usually need to address:

- sediment and substrate characteristics;

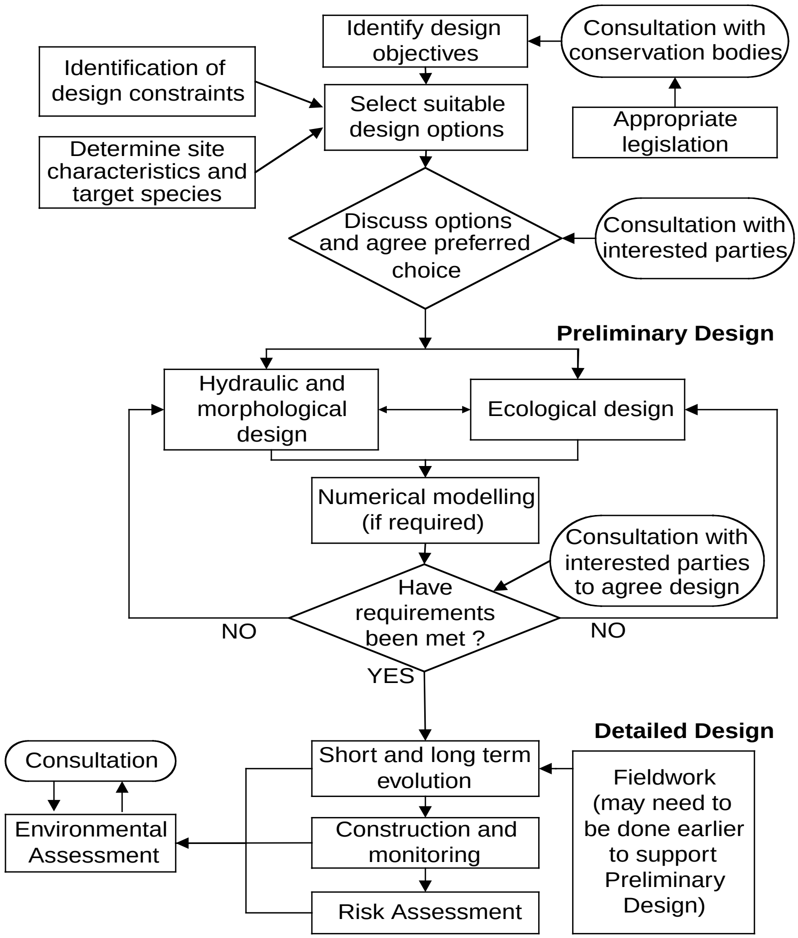

Fig. 4.1 Design flow chart for managed realignment.

- hydrodynamic, hydrological and morphological characteristics;
- habitat and species characteristics

as outlined in Table 4.3 [see ABP Research (1997) for further elaboration; available at www.estuary-guide.net].

The process of designing a scheme, based on all the above considerations, is most effectively begun with a walk-over of the site and an interrogation of the topographic maps of the site (e.g. using LiDAR survey data). Topographic maps can be used to identify features such relic creeks in historically reclaimed land and these can be the best guide to designing a scheme because they betray the natural morphology and thus indicate the best approach to achieving habitat restoration. Carrying out a site visit with this topographic data, both at high and low water, is also valuable for reviewing sea wall integrity, identifying breach locations (e.g. ideally at eroding locations/promontories and positions where sluices already exist), and assessing patterns of erosion and accretion in the adjacent estuary/coast to identify whether the site needs to be realigned in discrete and hydrodynamically separate phases.

Table 4.3 Indicative scope for site characterization

Sediment and substrate	Hydrodynamic and hydrological	Habitats and species
Type of strata and sediment characteristics including: • particle size analysis • bedrock/boulder clay levels • organic carbon • shear strength for cohesive soils • sediment density tests Resuspension parameters for nearby intertidal mudflats Sediment consolidation Sediment quality Type and quantity of sediment material deposited seasonally Levels of suspended sediments in the adjacent areas Imported material characteristics (if required)	Tidal characteristics (levels, asymmetry, discharge/prism) Channel and intertidal flow velocities Morphological parameters (hydraulic geometry, hypsometry) Location of aquifers and discharges, and volumes of freshwater input Groundwater levels Water chemistry Water quality including nutrients, bacteria, heavy metals, polyaromatic hydrocarbons and pesticides	Nutrient and organic carbon accretion rates seasonally Monitoring of existing usage and identification of requirements for: Benthic microbes Phytoplankton Benthos Fish (adult, juveniles, spawning grounds) Vegetation Weed species identification Overwintering birds Bird feeding/migration Patterns of feeding Need for buffer zones

Developing the design layout

The preliminary design is about exploring the art of the possible, seeking to find a suitable combination of components to deliver the desired habitats and related scheme objectives, whilst working within the identified constraints. This invariably gives rise to conflicting requirements; some of these can be addressed by optimizing the options but others will need to be resolved by suitable compromise. So, for example, whilst it may be desirable to maximize the tidal prism within the site to achieve the desired range of habitats, there may be a conflicting requirement to constrain the prism to a certain magnitude in order to avoid undue impacts outside the site. Determining the optimum balance invariably involves consideration of a range of options and a process of iteration to develop a scheme that is acceptable. There are a number of options that form the basis of the design, either individually or in some suitable combination. These can be described in terms of their form, habitat type or both. A number of options are summarized in Table 4.4.

Managed realignment can be used in one of two ways. The most common use, as has been described previously, is as a means of creating, or reinstating, habitat within a particular site. The alternative is to use managed realignment as a means of improving the resilience of an estuary. This can be done by selecting the location of the site(s) to enhance the dissipation of the tidal wave and by increasing the accommodation space (the area the estuary occupies as it migrates landwards in response to sea level rise) by removing rather than simply breaching the sea walls (Townend and Pethick 2002). In some cases the latter may need to be done in stages; first breaching the walls to allow the site to accrete to a level compatible with the external mudflat or saltmarsh and then, some time later, removing the remaining sea wall (or allowing it to simply collapse over time).

The nature of the openings to the site will depend on the type of habitat to be created. This can range from the complete removal of the existing sea wall, as just noted, through the formation of one, or more, breaches, to the use of some form of structure such as culverts, weirs and sluices. Standard texts on hydraulic engineering provide details on the design of culverts, weirs and sluices (Chow 1959), and a method for designing a breach in a sea wall is given in Townend (2008).

Within the site, there is invariably a need to provide some means for the tide to propagate into the area. Where the chosen option requires marine

Table 4.4 Summary of design options

Feature	Concept	Suitable for
Intertidal mudflat or sandflat	Low-lying sites that can be opened up to a high degree of exposure will tend to form mudflat or sandflat depending on the sedimentary environment	Benthic fauna and feeding resources for overwintering bird communities and fish species
Saltmarsh	Can form at the back of a mudflat, or in a sheltered breached enclosure with an elevation of MHWN and above. A suitable dendritic channel network will be required to provide tidal inundation	Saltmarsh and possibly grazing marsh. Also fish feeding areas
Saline lagoon	Use some form of regulated exchange to allow site to be inundated with salt water and limit the release so that water is retained in the lagoon	Specialist BAP invertebrate habitat and wildfowl roosting/feeding area
High marsh lagoon	Shallow 'pans' created in upper saltmarsh that act like saline lagoons	Specialist BAP invertebrate habitat and wildfowl roosting/feeding area
Tidal creek	Large-scale channels, with associated mudflat and saltmarsh, and possibly a dendritic network of tributary channels	Subtidal, intertidal and saltmarsh fauna. Also fish feeding areas
Intertidal islands	Areas of high ground (above high water) within the intertidal provide safe areas for bird roosts	Bird roosting and breeding sites as well as locations for plant and terrestrial invertebrate species
Brackish marsh and lagoon	Similar to the saline lagoon but using freshwater sources	Mitigation habitat for losses incurred to invertebrates and water voles on the island
Freshwater marsh	Can be created above the tidal frame with a suitable freshwater supply	Freshwater marsh and grazing marsh
Regulated tidal exchange	A combination of weirs to allow the top of the tide to enter a section of the site, with one-way sluices to release the water at low tide	Mixed brackish wetland habitat dependent on land morphology and water exchange volumes

BAP, biodiversity action plan; MHWN, mean high water neaps.

inundation, it is often necessary to provide a series of channels or ditches to promote both influx and drainage of the tidal waters. In many instances it may be appropriate to simply modify the existing field drains to establish a coherent network. This may not be ideal from a morphological point of view but, by using what is there, the amount of earthworks and hence costs are minimized (Figs. 4.2 and 4.3). In sites without extensive field drains, it is necessary to develop a dendritic channel network that fits within the planned topography for the site. For sites that have been historically reclaimed and are being reinstated, it is often possible to identify the original channel network on aerial photographs. This is because the infilling and subsequent differential settlement of the original channels produces minor differences in level and drainage across the surface that are visible in vertical aerial photographs. These can be used as a template for the channels in the reinstated site.

However, in some sites it is necessary to develop a channel layout based on the site topography and the habitats that are to be supported (Fig. 4.4; see also Fig. 4.2). Various guidance is available on the hydraulic and morphological geometry required for successful habitat restoration schemes (Krone 1993; French 1996; Haltiner *et al.* 1997; Friedrichs and Perry 2001; Williams *et al.* 2002). In addition, a number of models are beginning to be developed that are able to represent the evolution of saltmarshes and mudflats and provide a more process-based means of establishing how creek networks (Marani *et al.* 2003; D'Alpaos *et al.* 2005; Temmerman *et al.* 2005; Hood 2007) and saltmarshes (Mudd *et al.* 2004; Morris 2006; D'Alpaos *et al.* 2007; Marani *et al.* 2007; Kirwan and Murray 2007; Temmerman *et al.* 2007) are likely to develop.

For preliminary design of the site, some simple rules can be used to explore the mix of marsh, mudflat and channel that may be possible on a

Fig. 4.2 Vertical aerial view of the Wallasea Island site on the east coast of the UK.

given site. The main functions of creek channels (French 1996) are to:
- supply and disperse fine sediment;
- provide efficient drainage and de-watering of sediments;
- dissipate tidal energy inputs.

The major creek channels within the site can be designed using empirical relationships between cross-sectional area and tidal prism or discharge (as proposed by O'Brien 1931; Myrick and Leopold 1963). For creeks within saltmarshes this process can be reduced to consideration of the plan area by

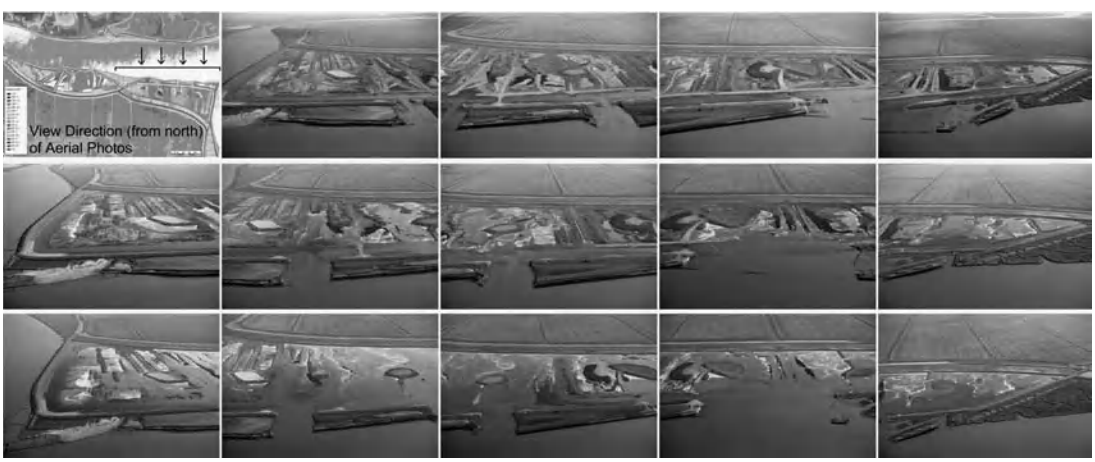

Fig. 4.3 Initial flooding of the first scheme on the Wallasea site, where the design made use of the existing field drains. (See the colour version of this figure in Colour Plate section.)

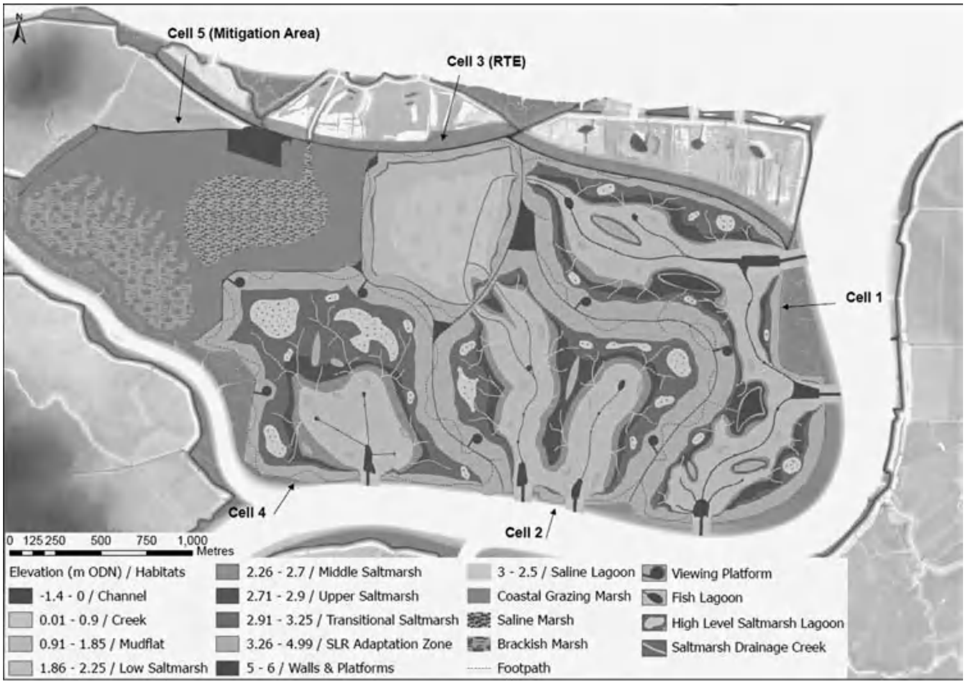

Fig. 4.4 Example of dendritic network design for a more extensive realignment on the Wallasea site. (See the Colour Version of this figure in Colour Plate section.)

assuming that the marsh has an approximately constant depth relative to high water and that the velocities are approximately uniform, such that:

$$W = k \cdot \sqrt{\beta \cdot S^{\alpha}} \quad \text{where} \quad \beta = \frac{W}{h} \quad \text{and} \quad \alpha \approx 1 \tag{4.1}$$

In this expression W is the channel width, S is the surface drainage area and h is the average, or hydraulic, depth of the channel (D'Alpaos *et al*. 2005). Data from Venice lagoon suggest the aspect ratio, β, has a value of around 6 within saltmarshes, whereas much higher values are observed for tidal flat channels (Marani *et al*. 2002). In contrast, data for Stiffkey Marsh in Norfolk, UK, exhibit a variation in aspect ratio as a function of stream order, with values between 2 in the upper marsh, increasing to about 8 towards the marsh edge (Lawrence *et al*. 2004). The constant, k, reflects the particular characteristics of the area (exposure, sediment characteristics, flow velocities, etc.). There are not a lot of published data available to define this coefficient and an analysis of other marshes in the locality is the best way of obtaining a suitable value. Data presented by Lawrence *et al*. (2004) suggest a value of the order of $k \sim 0.01$, and data for Venice lagoon suggest a value of the order of $k \sim 0.005$ (Rinaldo *et al*. 1999).

Tidal asymmetry is an important consideration (Dronkers 2005), with ebb dominant channels tending to be found where the channels are narrow and there are extensive mudflats that are relatively high in the tidal frame. In contrast, flood dominance occurs when the channels are deep and wide and the flats are low in the tidal frame. It is generally considered better to create a flood-dominant system and allow some siltation of the channels and flats, rather than make the channels too small or the flats too high, as this requires the tidal action to erode the system and, depending on the consolidated nature of the site sediments, this may not be possible, or may take a very long time. In considering the movement of sediments in and out of the site and the extent to which the site is likely to be a source or a sink, it is important to consider the impact on the external estuary, although this may be most readily done as part of the detailed modelling phase. Modelling of the layout can also be used to check that the flow speeds are reasonably uniform across the site (and through the breaches). Typically these need to be in the range 0.5–0.7 $\mathrm{m\,s^{-1}}$ to sustain mudflat habitat – being sufficient to prevent accretion and potential for saltmarsh colonization but insufficient to erode the channel or flat.

Where possible, meanders should be introduced into the main creeks, as this reduces the available fetch and hence the influence of waves (Allen and Pye 1992). Subdivision of the area by a series of branching creeks further helps with dissipation of tidal energy and the reduction of wave propagation. It is also the most efficient way to both drain the site and provide a supply of water and sediment across the whole of the intertidal surface.

The appropriate junction angles can be based on the ratio of the tidal discharges within the branches (French 1996), or for preliminary design purposes it may be sufficient to set them at 120° where the branches are of equal size and at 90° where a minor tributary feeds into the main channel (Haltiner and Williams 1987). For the latter, there may also be a need to grade the bed of the channel into the main channel to avoid ponding.

Within saltmarsh, the additional friction due to the vegetation means that flow velocities slow and sediment is deposited. The ability of the flow to transport sediment across the marsh therefore decays with distance from the channels. Observation of the space-filling nature of dendritic creeks as well as work on the transport and deposition of sediment through the marsh canopy suggest that no point on the marsh should be more than 30–70 m from a creek channel (Haltiner and Williams 1987; Pethick 1994; Marani *et al.* 2003).

Recent developments in the understanding of saltmarsh behaviour and its interaction with the hydraulics and morphology make it possible to estimate the likely extent of saltmarsh and the associated sediment demand. Traditionally, saltmarsh extent has been defined using empirical equations relating the presence of particular species to the tidal range, hydroperiod (duration of submersion) and exposure (see paper by Grey in Allen and Pye 1992). More recent developments relate the behaviour to the available sediment supply, tidal conditions and biomass density (Morris *et al.* 2002; Temmerman *et al.* 2004). For preliminary design it is sufficient to know the likely species, the optimum depth below high water of the marsh community, and the maximum depth below high water that any of the species present is able to colonize.

The unvegetated slope for a mudflat can be developed using some form of idealized intertidal profile (Lee and Mehta 1995; Friedrichs and Aubrey 1996). A developed marsh will tend to intersect this profile around the maximum colonization depth and then form a step, rapidly grading up to the optimum marsh elevation. If the site is left to develop this transition naturally, it may give rise to a significant sediment demand.

The basis for developing a channel layout will be constrained by the size of the site, its length and breadth, the number of breaches, and the requirements that:

- the channel network meets the constraint defined by Equation 4.1; and
- that no point on the marsh is more than a defined distance from a channel.

The relative level of the intertidal alongside the tidal channels and its slope away from the channels, determines site hypsometry and hence the potential for saltmarsh colonization, allowing the extent of mudflat and saltmarsh to be mapped out. If the site is simply opened up to tidal influence, the hypsometry of the site will largely determine the distribution of habitats. Where specific habitat objectives have been set, it may be necessary to move material around the site, or import/export material to achieve the desired topographic variations and hence habitats.

A simple way of developing a layout, based on the foregoing requirements, is to place a simple grid over a plan of the site, with the grid spacing defined by the maximum distance to a channel

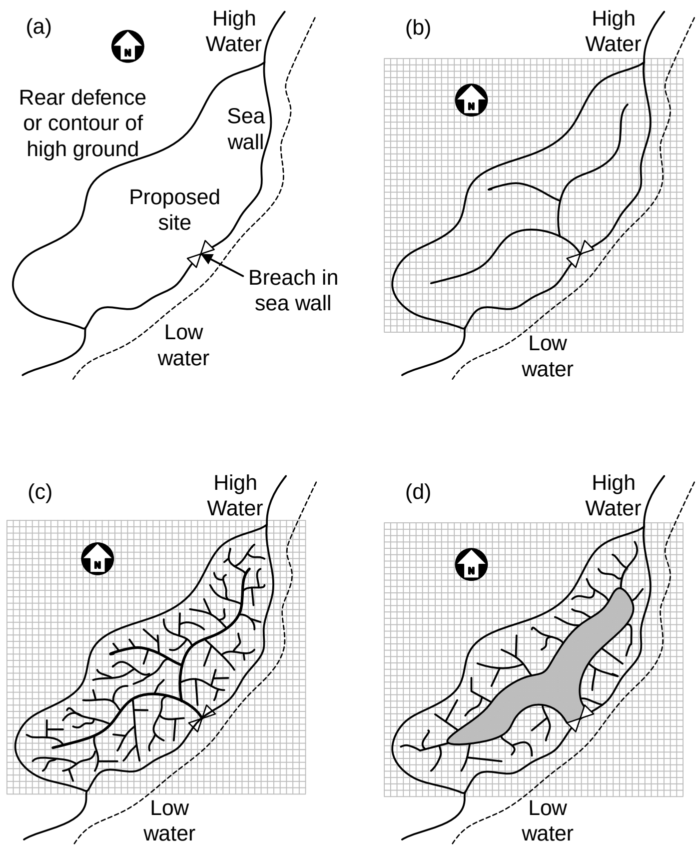

Fig. 4.5 Illustration of site layout design: (a) site plan; (b) base grid and main channel configuration; (c) marsh-dominated dendritic network; (d) mudflat and marsh creeks.

(Fig. 4.5). The main channels are then sketched on to reflect the shape of the site and any topography that might determine flow directions, or cut and fill volumes. Meanders and channel branches can be based on the simple guidance at this stage (see Fig. 4.5b). For a predominantly saltmarsh system, the dendritic network can then be developed around the main channels so that they provide water and sediment to every part of the marsh – with this graphical method any point on the marsh should be within a grid square (two grid squares at most) from a channel (Fig. 4.5c). The widths and depths of the channels can then be worked out using Equation 4.1. Where the site is lower in the tidal frame, part of the site will be mudflat. In this case, the marsh creeks will migrate into the marsh from the marsh edge (Fig. 4.5d), and a similar basis for spacing of the creeks can be adopted. The position of the marsh edge is determined by mapping the contour of the maximum colonization depth and assuming a step (or cliff) up to the optimum marsh elevation. As the area of mudflat increases so the site becomes more like a tidal inlet, and the sizing of the channels will be governed by the tidal prism, rather than the surface area as expressed in Equation 4.1. In this case, the size of the channels can be estimated using well-established prism or discharge relationships (O'Brien 1931; Myrick and Leopold 1963; Townend 2005). The worked example in Box 4.1 serves to further illustrate the process.

Whilst the foregoing focuses on how to establish site layouts to achieve the desired habitats, there are a number of other considerations that

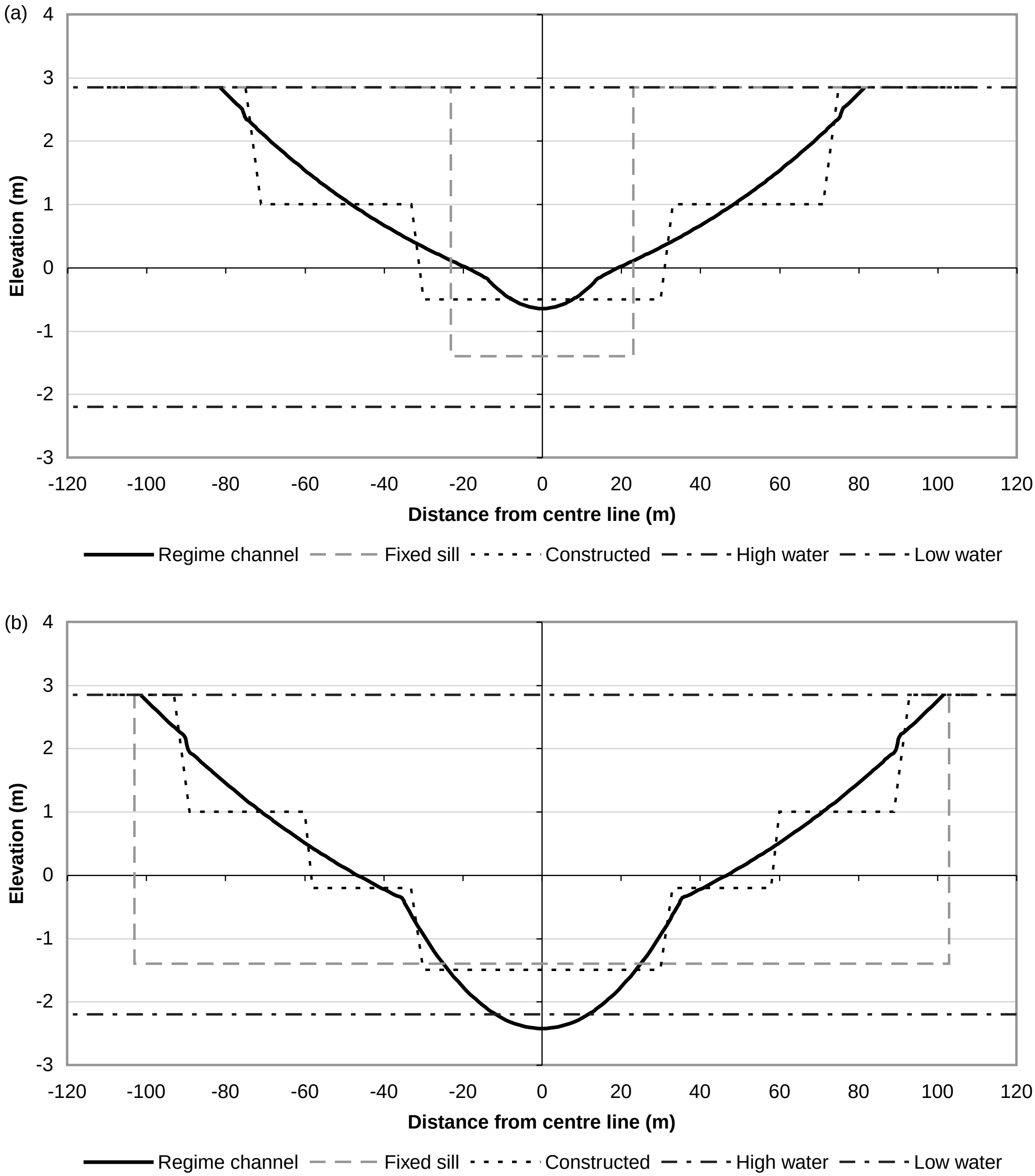

Fig. 4.6 (a) Resultant breach section for marsh site. (b) Resultant breach section for mudflat and marsh site.

Box 4.1 Worked example

A 186-ha site is enclosed by a fronting sea wall with high ground to landward. The sea wall is a clay embankment with 1:2 side slopes. The site itself slopes from a low point of 0.7 mODN (ODN = Ordnance Datum Newlyn) up to 3.5 mODN at the rear. The borrow ditch used to form the sea wall has an invert at −1.4 mODN.

The optimum depth for the marsh is 0.4 m below MHW (mean high water), i.e. about +2.0 mODN. The maximum depth of colonization is 0.6 m below MHW, i.e. +1.8 mODN.

Site data

MHWS = 2.85 mODN	*Area at HW = 186 ha*
MHWN = 1.90 mODN	Lowest level of site= −1.4 mODN
MLWN = −1.35 mODN	Bed density of mudflat = 1300 kg m^{-3}
MLWS = −2.20 mODN	

Target Habitats

Marsh-dominated dendritic network

Following the graphical method outlined above, an indicative network can be drawn as shown in Figure 4.5c. The prism at MHWS is simply the depth over the marsh × the area, i.e. 1.6 Mm^3 (Mm^3 = million m^3). Using Equation 4.1 to determine the section close to the breach, the width will be 16–22 m and the depth 2.5–3.0 m, depending on the values of k and β that are assumed. The upstream area can be used to determine the dimensions at any other point in the network.

Mudflat and marsh creeks

If an area of mudflat is to be created within the site this will need to be lower in the tidal frame. In Figure 4.5d, some 44 ha of the site is lower than +1.8 mODN and therefore likely to be mud. In this case, if we define 10 ha of the site as the area at the lowest elevation in the site (i.e. −1.4 mODN), there is a reduced prism over the marsh of 1.2 Mm^3 and an additional volume over the mudflat, which we assume slopes from the 10 ha at −1.4 mODN up to the 44 ha at +1.8 mODN. This results in an additional volume of about 1.3 Mm^3, so that the total tidal prism increases to some 2.5 Mm^3.

Given the existing site topography and the very different design topographies of the two schemes (cf. prisms of 1.6 and 2.5 Mm^3) there is likely to be very different cut and fill requirements and quite possibly the need to import material for the marsh-dominated scheme. The exact earthwork requirements will need more detailed work but these initial estimates do allow habitat aspirations to be quickly related to construction implications.

The size of the breach (or breaches) can be estimated using the method outlined in Townend (2008) and the resultant breach sections are illustrated in Figure 4.6. The 'constructed' line in the figures shows how this section is likely to be constructed, with a series of working platforms, allowing the tidal flow to do the final trimming. However, it must be remembered that whilst some preliminary lowering can take place, the final opening generally has to take place over a single tide. This in itself limits the size of breach that can be constructed and, in many cases, leads to a preference for several smaller breaches than the single large breach shown here.

(MHWS, mean high water springs; MHWN, mean high water neaps; MLWS, mean low water springs; MLWN, mean low water neaps.)

need to be worked through in order to develop a robust design. The sort of questions that have to be addressed include:

• How will the site change the hydraulic and sedimentary regime in the area adjacent to the site?
• Is there sufficient flushing of the site?
• Will the breach cause channel incisions in the existing mudflat?
• Will the flows in and out of the site disturb other interests (navigation, recreation, shell fisheries, etc.)?
• Is there any contamination in any of the sediments that are likely to be disturbed?
• What is the balance of saline and freshwater in the site and is this appropriate for the target habitats?
• What is the potential impact on offsite flood hazards and drainage?
• Is there any requirement to include public access or other community benefits on the site?
• Is there any threat from invasive species?

The full range of issues to be considered is addressed in the Habitat Review (ABP Research 1998, available at www.estuary-guide.net) and in various publications (Zedler 2000; Zedler & Callaway 2001; Williams and Faber 2004; Leggett *et al.* 2004, and a number of websites – see 'Online resources' below).

Once an outline layout has been established, this can be modelled and the layout refined to achieve the desired performance. This will usually entail the use of well-established models to examine water flows, sediment transport and morphological change (Abbott and Price 1994; Reeve *et al.* 2004). To make an assessment of the long-term development of the site and its impact on the surrounding area a hybrid model (Huthnance *et al.* 2007) such as the regime model (Wright and Townend, 2006; Spearman 2007) or simplified box models such as ASMITA (Aggregated Scale Morphological Interaction between a Tidal Basin and the Adjacent coast; Stive *et al.* 1998) can be used. As already noted, a number of models are now becoming available that can represent the evolution of creek networks, saltmarshes and mudflats (Marani *et al.* 2003, 2007; D'Alpaos *et al.* 2005, 2007; Mudd *et al.* 2004; Morris 2006; Hood 2007; Kirwan and Murray 2007), and detailed process models are being extended to include biological processes such as biostabilizers, bioturbators and vegetation such as saltmarsh (Widdows *et al.* 2000).

Finally, in developing scheme proposals it is important to be aware of the uncertainties inherent in the design process, particularly relating to habitat rate of development and extent, to manage expectations and to be pragmatic about the targets set. In particular, net sediment accretion and the resulting increase in bed levels is a common occurrence in realignment sites although the rates vary depending on the sediment supply and the hydrodynamic conditions. For instance, in the very turbid Humber Estuary, accretion has led to clear changes in habitat from mudflat to saltmarsh over a relatively short timeframe (e.g. at Paul Holme Strays and Welwick). In those instances where habitat delivery targets are necessary or requested then they can readily be framed to anticipate and accept the likelihood of such changes (ABPmer 2004b). Similarly, there is an increasing recognition that the best solution is the one that uses the natural morphology to best effect and that physical manipulation of the landform to achieve a predetermined habitat mosaic is usually not warranted (beyond that required to achieve a hydrodynamically stable system). Such intervention is not only unlikely to achieve a fixed habitat composition but also might not be feasible before realignment because of cost and sediment management implications, and is almost certainly impossible after realignment due to safety and access issues. Indeed, some useful features can often be formed, without directed intervention, from the site's natural morphological complexity. At the Wallasea North Bank realignment site both mudflat and saltmarsh were created as compensation targets, and these habitats currently appear to be functioning well. However, there are also areas of intermediate elevation (e.g. areas of excavated clay spoil deposition on the mud) that are poorly vegetated because they are too low in the tidal frame and have a paucity of invertebrates because of their elevation and sediment type.

Before the breach there was no reason or requirement to design these in but, even though they were not a target habitat, they have provided additional roost sites for waders and so form part of the habitat mosaic within the site (Jacobs/ABPmer 2008).

Obtaining Planning Approval and Consents

The effective implementation of a successful realignment scheme relies on a combination of forward planning, good project management and effective consultations (d'Herbemont & César 1998).

While the full list of consents and licences that are needed for realignments varies between projects due to differences in site-specific conditions and site location, a good understanding of the key requirements is provided from reviews of past project case examples and generic guidance (Nottage and Robertson 2005; Dixon *et al.* 2008; and see websites listed under 'Online resources' below). Based on this past experience the key consent and licence requirements, in England and Wales, can include the following:

• Town and Country Planning Act (1990) and Town and Country Planning Act, Environmental Impact Assessment Regulations 1999 (the EIA Regulations) – planning permission required from the local authority for which an EIA is likely to be required to accompany the planning application.

• Land Drainage Act (1991) – consent needed from the Environment Agency regarding changes to land drainage.

• Water Resources Act (1991) – consent required from the Environment Agency Flood Defence Committee for proposed works affecting tidal flood defences for which a Flood Risk Assessment may need to be submitted with planning application. Under the same Act, discharge consent may be required where there will a discharge from the site to the estuary/coast.

• Wildlife and Countryside Act 1981, as amended by the Countryside and Rights of Way Act (CRoW) 2000 – under advice of conservation agencies may need to include assessments of impacts to species that are protected either under Section or under the Habitats Regulations.

• Habitats Regulations (1994) – where impacts to a European marine site could be significant an Appropriate Assessment may be required (information required for this assessment should be provided in the Environmental Statement).

• Coast Protection Act (1949) consent or Works Licence from Harbour Authority – this is needed to address issues relating to the impacts on navigation below the high water mark.

• Crown Estates consultation and consent – required to safeguard land ownership (otherwise new coastal habitat could revert to the Crown after breaching).

• Highways Act 1980 or the Town and Country Planning Act 1990 – a footpath diversion order may be required where an established public footpath exists.

• Waste Management Licensing Regulations 1994 – where sediment is to be imported a waste management licence or an exemption (e.g. sediment volume $< 20{,}000\,m^3$). Importantly, for realignment projects where the imported sediment is manifestly designed to achieve a useful purpose or 'recovery' then this waste/fill material deposit is not liable for landfill tax.

• Food and Environmental Protection Act (1985) – a FEPA licence for the deposit of materials in the marine environment, or construction below high water may be needed. However, in most cases any deposits of material will be below the future, not the existing mean high water mark and will not require a licence. Notwithstanding this, it is best practice to adhere to FEPA-standard quality requirements for the marine deposition of sediments in order to allay concerns and avoid impacts arising from the import of contaminated materials.

The timing of the consenting and construction windows are often linked (Dixon *et al.* 2008). Typical timescales for some of the key tasks are as follows:

• Site investigations, project design and EIA preparation – 12 months.

• Securing planning permission and other consents – 12 months.

• Major coastal defence earthworks – 3 months per km.
• Placement of dredged material – 4 months in two tranches over the winter (November/December and February/March).
• Settlement, consolidation and vegetation of walls – 12 months ideally but can be accelerated by engineering.
• Breaching of the sea walls – 2 weeks in two separate tranches between the top of spring tide and bottom of neap tide.
• Post-breach monitoring – 5 years to validate hydrodynamic predictions and assess attainment of compensation targets.

In viewing the whole project plan, one of the critical potential obstacles or 'pinch points' that is largely outwith project management control is the response of local people and politicians. This aspect can be mitigated by adopting comprehensive and early consultations with a wide range of groups and individuals including the general public, statutory authorities, specialist interest groups and estuary users (ComCoast 2007). Not only is this very important for communicating the need for, and rationale of, the project it also ensures that interested parties (especially locals) feel involved in the process and can lead to extra components being included in the scheme design (e.g. a recreational area inside the site and signage on the new sea walls). This process is also valuable for informing the Consent and EIA process by enabling key issues to be highlighted.

Undertaking the Construction Work

For the most part the construction work for realignment requires the same plant and approach as would be expected for any typical infrastructure development. The amount and type of plant will clearly be influenced by the location, scale and timescales of the work and, as described above, there needs to be a clear timetable of events to achieve the required tasks and meet important deadlines. Deadlines can include engineering ones, such as ensuring that all land works are completed in time for the smaller midsummer tides that are best for breaching, or they can be environmental, such as making sure land is 'sterilized' before the spring and early summer bird breeding season.

One of the absolutely critical considerations in any construction exercise, but one that brings its own unique issues for those undertaking realignments, is health and safety. The building in of health and safety considerations as part of the project planning is a legal requirement [e.g. The Construction (Design & Management) Regulations 2007, or CDM 2007], and on realignments the sort of considerations that are relevant include:
• Building in health and safety requirements into the scheme's morphological design. This can include making sure that there are safe access routes for plant into and out of the site. Larger sites may need to be divided into discrete sections or 'cells' so that not all the breaching has to be done in one go (i.e. in one tidal window).
• Timing the breach construction to be carried out on a neap tide (so that flow speeds through the breach will be at their lowest). This needs to be planned with a clear understanding of how much excavation can be undertaken in the low-tide window. For the final stage breaching at the Wallasea North Bank site, a 330-m length of sea wall material was removed at three breach locations during a single 7-hour tidal window, and this is thought to be about the maximum that can be achieved safely. The amount of work required on the final breach can be minimized by slowly stripping down the walls, as the tide levels drop from a spring to a neap tide.
• Ensuring that all workers have appropriate Personal Protective Equipment (PPE) such as life-jackets as well as having stand-by plant that is not required but can be brought into service if there are any problems with the plant being used.
• Safe methods of working, such as pinning open the digger doors during breaching to allow drivers to exit easily, and deployment of trained shoreline support staff who have throw lines and other equipment in the event of any problems.
• A clear project management structure with one leading project manager to ensure that there is no mixing of messages to the plant operators.

Monitoring and Evaluating Scheme Performance

A final key component of a successful realignment is the implementation of an effective monitoring programme. This has two key functions: to verify the impact predictions and to assess the site's development (e.g. against compensation or biodiversity targets). Generic guidance on monitoring methods is available (Defra/EA 2004).

As with many other aspects of managed realignment though, the detailed composition of the monitoring programme will reflect site-specific requirements. For instance, while Wallasea North Bank needed careful consideration of the hydrodynamic impacts within the Crouch Estuary (due to the 2% tidal prism change) this was not relevant for a site in the Humber Estuary (also on the east coast of the UK), where the tidal prism change is negligible (ABPmer 2004b). Table 4.5 summarizes the types of monitoring adopted over a range of different UK projects (site locations are shown in Fig. 4.7).

As part of the process of designing bespoke monitoring programmes it is recommended that careful consideration is given to the methodology and the value of the information in the context, especially, of the costs that will be incurred for its collection. For instance, taking and analysing benthic invertebrate samples according to standardized quantitative methods can be very costly when, for the purposes of broad-scale site evaluations, all that may be needed is a qualitative survey of community status to provide an indication of ecological functionality and waterbird prey resource levels. Thus the importance of the information must be established and a clear dissociation maintained between what is essential and what is 'nice to know'.

Table 4.5 Types of monitoring undertaken at recent UK managed realignment sites

Site No.	Site	Accretion/ Erosion on site	Accretion/ Erosion off site	Bathymetry	Tidal levels and/or flow velocities	Invertebrates	Vegetation	Birds	Fish	Soil and/or sediment quality
1	Northey Island	x					x			
3	Orplands	x					x			x
4	Tollesbury	x	x	x		x	x	x		x
6	Lantern Marsh	x					x			
8	Trimley					x	x	x		x
10	Pillmouth									x
11	Bleadon Levels	x				x	x	x		
12	Abbotts Hall	x	x	x	X				x	
13	Brancaster West Marsh						x			
14	Brandy Hole						x			
15	Freiston Shore	x	x	x	x	x	x	x	x	
16	Nigg Bay	x				x	x	x		
17	Paull Holme Strays	x				x	x	x		
19	Alkborough			x		x	x	x		
21	Chowder Ness	x		x		x		x		
22	Wallasea	x	x	x	x	x	x	x		
23	Welwick	x		x		x		x		

Source: http://www.abpmer.net/omreg/; see Fig. 4.7 for site locations.
N.B. No or very limited monitoring undertaken at the following UK sites: Annery Kiln, Black Devon Wetlands, Thorness Bay, Watertown Farm. Details are unknown for the following UK sites: Alnmouth, Cone Pill, Glasson, Horsey Island, Millennium Terraces, Montrose Basin, Pawlett Hams.

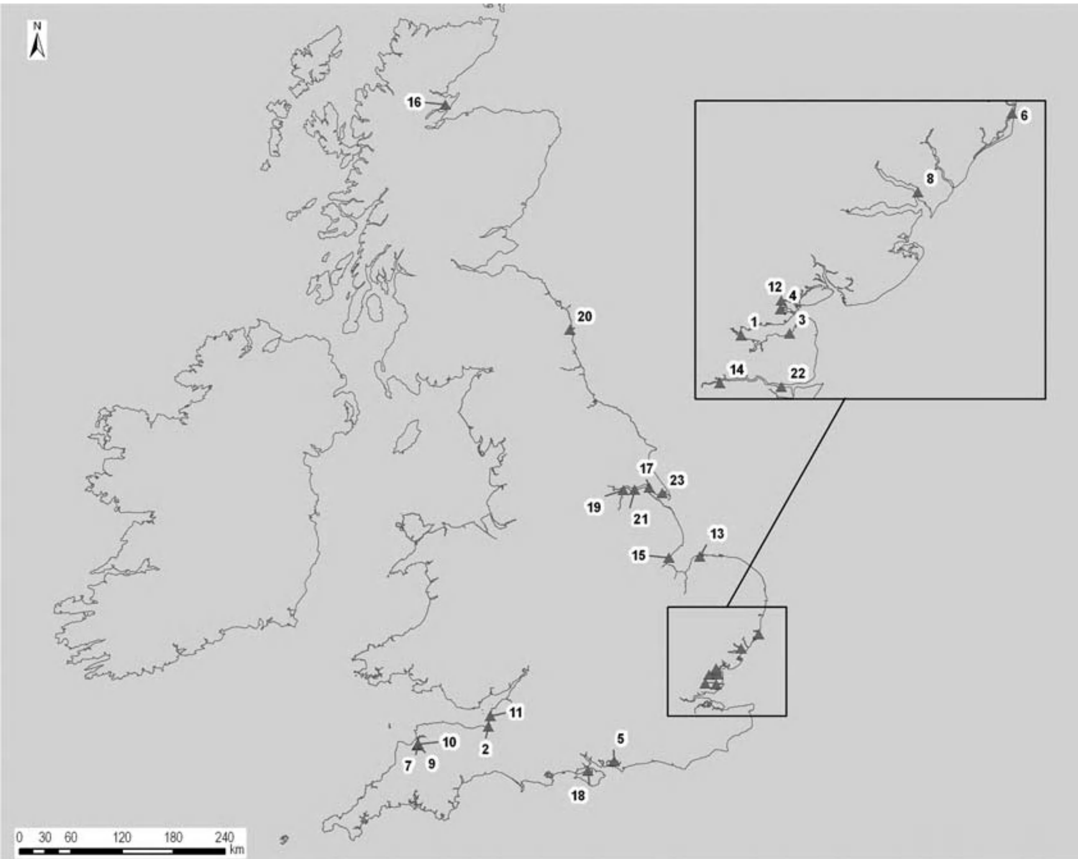

Fig. 4.7 Location of recent UK managed realignment sites, as detailed in Table 4.5.

After breaching, and as part of the monitoring work, it is strongly recommended that communication, which was vital before and during the scheme's implementation, is continued afterwards. This can be achieved through the circulation of annual newsletters, discussion papers, and details of specialist site visits. Furthermore, on-site information boards help to inform the public about why the wetlands have been created.

The dissemination of monitoring results and the lessons learned from past schemes is also vital in helping to improve future managed realignments. This includes the circulation of findings from individual projects as well as initiatives such as the development of an online database of managed realignments (Rupp-Armstrong *et al.* 2008). Such database resources provide a way of auditing progress, both nationally and internationally, and help to demonstrate to the community how effective schemes have been. This, in turn, will improve the quality of future realignment schemes as well as the confidence that the coastal management community and the general public can have in these projects.

Conclusions

Over the last 20 years, some 76 Managed Realignments have been completed across northern

Europe and, while the primary drivers for these schemes have varied greatly (nature conservation, coastal adaptation and project/plan compensation), the problems encountered, such as public opposition and planning delays, have often been very similar (Rupp-Armstrong *et al*. 2008). In particular, the most common lesson identified by implementers is that the public and stakeholders should be engaged early in the process. There is no doubt that projects are facilitated by such early engagement or, equally, that projects can stall where the engagement either hasn't been pursued efficiently enough and/or where the rationale for the initiative has not been effectively conveyed.

This aspect is now widely recognized by those involved in progressing managed realignments; also a vast amount is now known about how best to implement such schemes by virtue of the practical experience that has been gained. As a result there has been a clear shift since the late 1980s from the early ad hoc barely modelled and monitored schemes to a new generation of sophisticated multi-driver schemes, which tend to be meticulously planned, consulted on and monitored. As one major example of where this road has led, in July 2009 the RSPB secured planning consent for a new 677-ha Wallasea Island Wild Coast Project. Located alongside the Wallasea North Bank site that was breached in 2006, this new, much larger, scheme will involve the importation of 7.5 million m^3 of inert fill materials to raise, restore and reform a sunken and flat arable landscape and create a site that alleviates the existing flood risk, integrates into the existing estuary hydrodynamics, and delivers large areas of new coastal habitat and recreational opportunities for the public. It is envisaged that the lessons learned from this new generation of more sophisticated large-scale schemes (alongside the lessons for past, present and future smaller-scale projects) will only help to further accelerate the 'learning by doing' process and, thus, improve the quality and value of future schemes for nature and people. Alongside such progress there will be a continuing need for statutory guidance at a national and regional level that can then be used to underpin projects at a local level. In doing so, this will help project mangers communicate the project rationale as clearly as possible to the members of the public that live in the vicinity of a site and are most closely influenced by, and most likely to benefit from, such schemes.

Online Resources

Sites providing guidance, case studies and useful sources of information (all accessed April 2010):

Wikipedia article on 'Managed retreat': http://en.wikipedia.org/wiki/Managed_retreat

Environment Agency Managed Realignment Electronic Platform: http://www.intertidalmanagement.co.uk/contents/index.htm

Online Managed Realignment Guide: http://www.abpmer.net/omreg/

Defra's Saltmarsh Management Manual: http://www.saltmarshmanagementmanual.co.uk/

The Estuary Guide from Defra/Environment Agency: http://www.estuary-guide.net/

ComCoast website: http://www.comcoast.org/

Wallasea Wetlands Creation Project: http://www.abpmer.net/wallasea/

See also: http://www.defra.gov.uk/rural/protected/wallasea.htm

UK Biodiversity Action Plan: http://www.ukbap.org.uk/GenPageText.aspx?id=98.

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5 Accounting for Sediment in Flood Risk Management

COLIN THORNE, NICK WALLERSTEIN, PHILIP SOAR, ANDREW BROOKES, DUNCAN WISHART, DAVID BIEDENHARN, STANFORD GIBSON, CHARLES LITTLE JR, DAVID MOONEY, CHESTER C. WATSON, TONY GREEN, TOM COULTHARD AND MARCO VAN DE WIEL

Abstract

River sediment dynamics at the catchment scale must be taken into account when managing flood risk sustainably because the presence and movement of sediment has important impacts both on the capacity of the channel to convey floods and the range and quality of habitats that it provides. However, conventional sediment transport calculations performed at the cross-sectional or reach scales cannot provide the information needed to manage sediment at the system or catchment scales. Recognizing this, a work package in Phase 1 of the Flood Risk Management Research Consortium (FRMRC) assembled a toolbox of methods and models capable of investigating broad-scale sediment transfer through the river system. The six approaches in the FRMRC Sediment Toolbox are: (i) Stream Power Screening Tool; (ii) River Energy Audit Scheme (REAS); (iii) Sediment Impact Analysis Method, embedded in HEC-RAS (HEC-RAS/SIAM); (iv) Hydrologic Engineering Center, River Analysis System (HEC-RAS version 4.0); (v) iSIS Sediment; and (vi) Cellular Automaton Evolutionary Slope and River Model (CAESAR). The simpler tools may be used to classify reaches as sediment sources (eroding reaches), pathways (dynamic equilibrium reaches) and sinks (depositional reaches) and construct reach- and catchment-scale budgets for stream power or sediment movement. The more advanced models can generate indicative data on rates of net erosion and deposition in unstable reaches. All of the methods and models have a role in the assessment, modelling and management of sediment-related flood risk. The tools are also useful in developing an understanding of long-stream connectivity in the fluvial system, which is important to the functioning and management of the river as an ecosystem.

Background

Context

Historically, flood risk management in the UK has seldom taken into account the erosion, transfer and deposition of sediment, or the effects of flood management on sediment dynamics in the fluvial system (Environment Agency 1998). However, schemes that disrupt sediment continuity or connectivity tend to require heavier and more frequent maintenance, either to prevent sedimentation from compromising the design capacity of the channel for flood conveyance, or to prevent deterioration or failure of flood defence assets due to fluvial erosion (HR Wallingford 2008). Also, the

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sediment impacts of a flood alleviation scheme may trigger channel instability elsewhere in the fluvial system (Sear *et al.* 2010). In this respect, recent research suggests that the effects of, for example, removing sediment by dredging for flood defence purposes may be much more damaging than previously realized (Wishart *et al.* 2008).

Currently, the standard method for investigation of catchment-scale sediment dynamics in UK rivers is the 'Fluvial Audit' (Thorne *et al.* 2010). In this approach, field and documentary investigations are used to divide the fluvial system into geomorphic reaches designated as sediment source (scouring), sediment transfer (dynamic equilibrium) or sediment sink (depositional) reaches. The approach rests on detailed field reconnaissance of the entire drainage network by experienced fluvial geomorphologists (Thorne 1998). The Fluvial Audit has proven very useful in river conservation and restoration projects, but it does not yield the quantification of the sediment dynamics required to interface effectively with the engineering components of strategic flood risk assessments, Catchment Flood Management Plans (CFMPs) or River Basin Management Plans (RBMPs).

Also, the insights into catchment sediment dynamics provided by a Fluvial Audit are reliant on accurate interpretation of field and archive evidence, which is always equivocal, by an experienced geomorphologist. Consequently, the outcomes are necessarily related to the quality of the expert judgement exercised by the person responsible for the investigation. However, the main limitation of the Fluvial Audit is that it has no inherent predictive capacity and so cannot simulate system response to different, proposed flood risk management actions – reducing its utility in options appraisal. The Fluvial Audit is now evolving to incorporate a modelling dimension (Thorne *et al.* 2010), but finding a quantitative approach to representing sediment dynamics that can be routinely applied across a range of catchment scales remains a difficult challenge that is yet to be solved in the context of practical applications.

Recognizing the practical limitations of existing, qualitative approaches, a component of the research pursued during Phase 1 of the Flood Risk Management Research Consortium (FRMRC) was directed at developing new tools to account for sediment in rivers, concentrating particularly on semi-quantitative and indicative characterization of the dynamics of the sediment transfer system at the catchment scale, and simulating system response to the impacts (intentional or unintentional) of existing and proposed interventions in the fluvial system that are related to flood risk management.

Computation of sediment movement is conventionally approached through application of the equations of fluid flow, sediment transport capacity and sediment continuity in a hydraulic or hydrodynamic model with a sediment module. However, the resources and field data required to apply these models restrict their use to the reach rather than the catchment scale, while extended run times mean that they cannot readily be used for the types of long-term simulation required to investigate sediment movement over protracted periods or through long reaches. Also, reliable sediment modelling demands both specialist training and prior experience on the part of the modeller, not only in hydraulic/hydrodynamic modelling, but also in the selection and appropriate use of different sediment transport equations.

It is in the context of the limitations of conventional qualitative and quantitative methods of accounting for sediment that the FRMRC assembled a toolbox of sediment methods and models that can be used by practitioners faced with the need to account for sediment or to solve sediment-related problems in flood risk management.

Strategically, use of the toolbox should aid understanding of the interactions between flood defence infrastructure and sediment dynamics, knowledge of which is vital in assessing the sediment impacts of existing flood alleviation schemes; appraising options during project planning and supporting detailed design of new schemes. Further, the toolbox also provides a means for end users to understand, and therefore account for, the impacts on flood defence infrastructure of changes in catchment sediment supply and adjustments of the fluvial system that seem ever more likely in the coming decades in response to climate change, changes in land use management and continued socioeconomic development in floodplains (Lane and Thorne 2008). Finally, the toolbox can support evaluation of the implications of infrastructure–sediment interactions for in-channel habitats and the ecosystems they support. The capability to link infrastructure, morphology, sediments and habitats when managing flood risk is essential in coordinating flood risk management and its operational delivery to the requirements of the Water Framework Directive and so relieving tensions that might otherwise develop between flood management and environmental legislation, to the detriment of river management practices that are holistic and sustainable (Lane and Thorne 2007, 2008).

Experience gained in using the toolbox will also provide a window on the future research needed to support improved understanding of sediment dynamics. The aim here is to help guide development of the next generation of broad-scale or whole-system flooding models by indicating how they might be made capable of recognizing and accounting for the impacts of sediment on future flood risks.

Selecting tools for the toolbox

From the outset, it was recognized that the need was to build on, rather than replace, qualitative assessments including the Fluvial Audit. In essence this called for selection of methods and models capable of providing an analytical basis for characterizing the sediment transfer system that minimized subjectivity and reduced reliance on expert judgement on the part of the user. On this basis, the methods and models selected for the toolbox were:

- Stream Power Screening Tool
- River Energy Audit Scheme (REAS)
- Sediment Impact Analysis Method, embedded in HEC-RAS (HEC-RAS/SIAM)
- Hydrologic Engineering Center River Analysis System (HEC-RAS version 4.0)
- iSIS Sediment
- Cellular Automaton Evolutionary Slope and River Model (CAESAR).

When investigating applications of these approaches, it soon became apparent that, due to the scarcity of sediment data for UK rivers and the complexity of sediment dynamics even in relatively simple watercourses, uncertainties in the results of quantitative, analytical sediment transport calculations would remain stubbornly high, so that the outcomes must customarily be labelled as 'indicative'. In light of this, elements of both analysis and interpretation remain essential to all sediment studies, regardless of the modelling tool employed, and the methods and models included in the toolbox all start by assuming that the user has a sound, qualitative understanding of the fluvial and sediment systems, gained from a Fluvial Audit or some equivalent methodology (Thorne *et al*. 2010).

While all the methods and models rely on elements of interpretation and analysis of the sediment transfer system, the relative contributions of these two components of scientific study vary between methods. In this regard, Figure 5.1 illustrates where each tool lies in the continuum between purely interpretive and fully analytical approaches.

Dealing with uncertainty

It is now accepted that uncertainty must be recognized and dealt with appropriately in all aspects of flood risk management, and accounting for sediment is no exception. No matter how sophisticated the analytical approach adopted, the fact is that sediment and fluvial processes cannot be perfectly observed or understood, let alone predicted. Uncertainty can be reduced through improved knowledge of the governing physical processes, by collecting sufficient data to represent the fluvial system and by ensuring that the model selected is properly designed, calibrated and verified. However, uncertainty due to natural variability is a property of the river, not the model, and it is to a degree irreducible. Once it has been accepted that some uncertainty is unavoidable, it is necessary to decide whether the existing level of uncertainty justifies the additional effort needed to reduce it further.

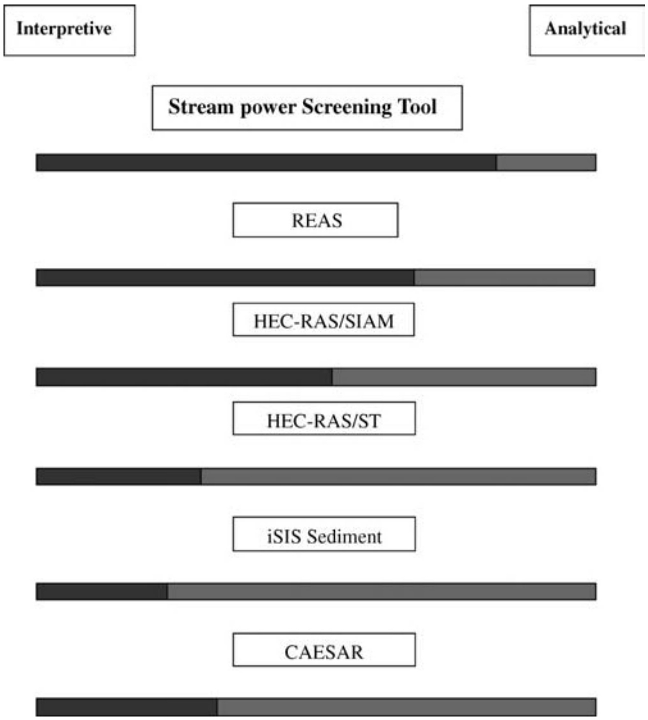

Fig. 5.1 Relative contributions of interpretational and analytical approaches in different tools in the FRMRC Sediment Toolbox. CAESAR, Cellular Automaton Evolutionary Slope and River Model; HEC-RAS Sediment Transport (ST), Hydrologic Engineering Center River Analysis System; REAS, River Energy Audit Scheme; SIAM, Sediment Impact Analysis Method.

In deciding whether further reductions in uncertainty merit the expenditure of time and resources necessary to achieve them, it is prudent to consider the level of risk stemming from the uncertainty. In some cases, uncertainties may be large, but may pose little risk to the outcome of a proposed project. Attempts to further reduce uncertainty through additional data collection or use of a higher accuracy model are justified only if the risk associated with the current level of uncertainty is too high to allow the project to proceed. In this context, according to Brookes and Dangerfield (2008), the level of uncertainty should be assessed as being:

- **Unacceptable:** risks associated with existing uncertainties cannot be tolerated and further effort must be made to reduce them at least to a tolerable level and ideally to an acceptable level.
- **Tolerable:** the risks associated with existing uncertainties are significant but can be lived with and effort to reduce uncertainties further would be disproportionate to the risk reductions achieved.
- **Acceptable:** the risks associated with existing uncertainties are insignificant.

It follows that, when selecting a sediment analysis tool and designing the data collection campaign to support it, the target should be to reduce uncertainties to the level at which they are acceptable or, at least, tolerable, recognizing that it is impossible to eliminate them entirely.

This approach requires consideration of **all** the various risks associated with sediment and its analysis, including those related to the nature of the problem being investigated, the physical environment, the objective of the modelling investigation, and the time, human resources and money available to support the work. The fact is that time, human resources and money will almost always limit the levels of detail and complexity that can be included in the study and the

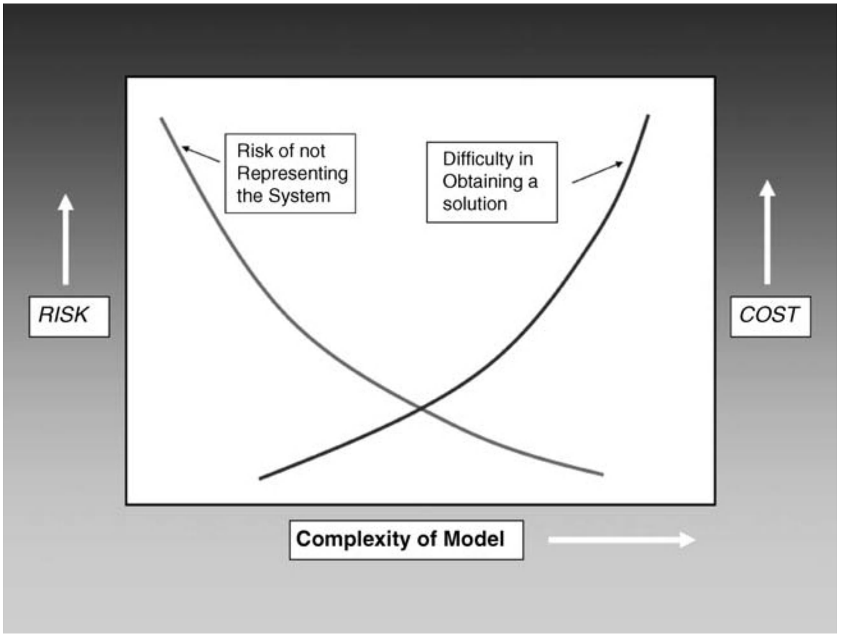

Fig. 5.2 Balancing cost and risk when selecting a model appropriate for a sediment study.

extent to which model calibration and validation can be performed. In this context, Overton and Meadows (1976) stated that:

> '. . .if a highly complex mathematical representation of the system under study is made, the risk of not representing the system will be minimised, but the difficulty of obtaining a meaningful solution will be maximised as much data will be required, and mathematical handling in a computer model (e.g. convergence and consistency) and complexity of mathematical processes may even render the problem formulation intractable. Further, the resource constraints on time, *(hu)*manpower and budgets may be exceeded. Conversely, if a greatly simplified model is selected the risk of not representing the physical system will be maximised, though the difficulty in obtaining a solution will be minimised.'

Careful consideration of the balance between the risks associated with uncertainty in representing the behaviour of the sediment system reliably and the rising costs associated with more complex models is therefore vital when selecting the level of model complexity appropriate to a particular sediment investigation and analysis (Fig. 5.2).

In assembling the methods and models to be included in the FRMRC Sediment Toolbox, the issues raised in the quotation from Overton and Meadows (1976) explain why a range of tools, extending from a very simple treatment of the sediment transfer system that is easy to apply (Stream Power Screening Tool) to an advanced, transport model that routes sediment by size fraction, but is costly, expertise intensive and time-consuming to apply (iSIS Sediment), was included. The premise here is that a risk-based approach to selecting the tool appropriate to the application is only possible provided that a suite of approaches that cover a wide range of complexities and funding situations is available to end users.

Stakeholder issues

The technical strength and veracity of a tool are two of a number of criteria that affect its utility and reputation amongst stakeholders and, hence, its uptake by end users. As explained earlier, a method or model will only be taken up widely if end users can support and apply it within the resource constraints imposed on time, expertise and funds – because the outcomes of a model are only as good as the interpretation of the numerical results. Even when technical end users recognize the need for complex or extensive (i.e. resource-intensive) sediment studies, they must still make a convincing case to other important and influential stakeholders, who may be reluctant to support such advanced modelling. For example, it may be difficult to persuade funders that the additional costs of advanced sediment assessment are justified, while politicians and the public are increasingly sceptical about the value of science-based approaches to understanding natural phenomena and will also only accept modelling outcomes that display the required blend of cognizance (stakeholders can understand and accept the model's principles and methods) and credibility (the model is not so simple or schematized that they scoff at it). These considerations are represented in Figure 5.3, which depicts the need to place the method or model within the central zone of a triangle of stakeholder requirements.

Issues of stakeholder attitude and end-user uptake were accounted for during assembly of the Sediment Toolbox through quarterly meetings with a steering panel that included representatives from both government agencies responsible for strategic and operational river management and consultants responsible for project-related sediment assessment, analysis and modelling. Consequently, the tools described in the Toolbox cover the range of each of the stakeholder-related criteria represented in Figure 5.3. In summary, the Toolbox is intended to provide a suite of tools that have the capability to assess sediment dynamics at the catchment scale, while making best use of available resources and achieving the required levels of uncertainty, technical complexity, risk, reliability, and stakeholder credibility.

The remainder of this chapter presents précis descriptions of each of the tools in the Toolbox, which should provide a sufficient basis from which end users and stakeholders can make initial judgements concerning the appropriateness of a particular method or model when selecting the approach or approaches to be adopted in a broad-scale sediment study. End users should then perform the background research and practical investigations necessary to apply the tool(s) selected for their particular application.

Stream Power Screening Tool

Basis and utility

In the context of the FRMRC Sediment Toolbox, development of the Stream Power Screening Tool was led by Brookes and Wishart (2006). The conceptual basis for the Stream Power Screening Tool

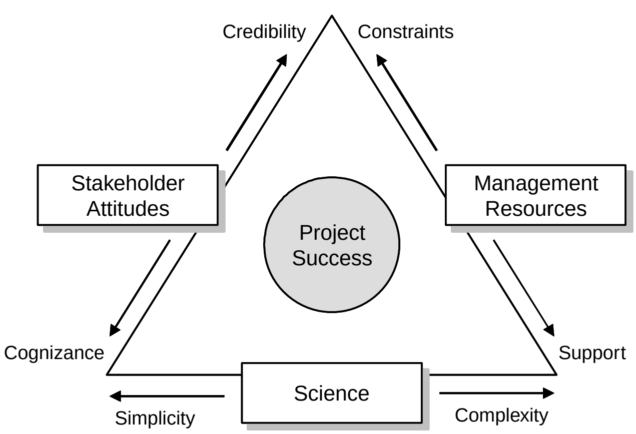

Fig. 5.3 Balancing management resources, the science base for sediment methods and models, and stakeholder attitudes to achieve project success.

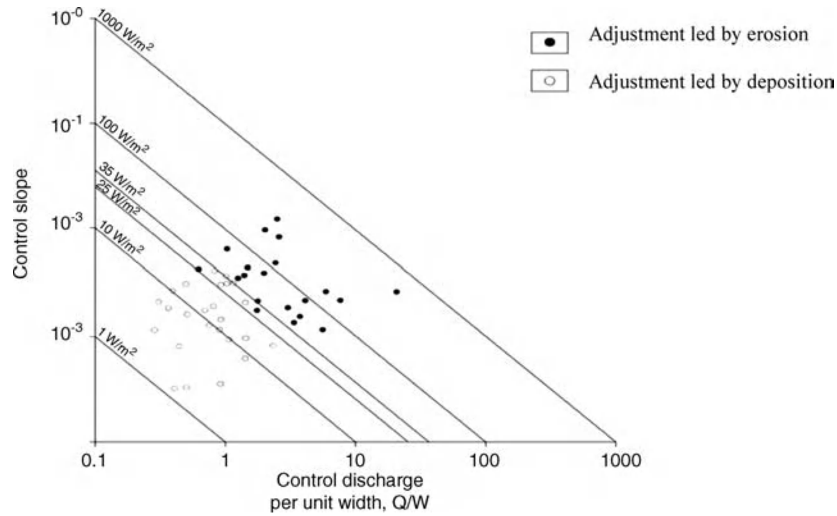

Fig. 5.4 Specific stream power plot showing channel response to straightening in several British rivers. Modified from Brookes and Wishart (2006).

springs from papers by Bagnold (1966, 1980). The most widely used index of stream power is the specific stream power, or stream power per unit area of the bed (ω), defined by:

$$\omega = \frac{\rho g Q S}{w} \tag{5.1}$$

where, ρ = water density (kg/m^3), g = acceleration due to gravity, Q = discharge (m^3/s), S = water surface slope and w = a representative channel width; the units of stream power are watts per square metre (W/m^2). This represents the stream power used by Bagnold (1966) in his sediment transport analyses, although he omitted the gravitational constant, g, in order that the units of specific stream power should match those of sediment transport per unit stream width (kg/m/s).

Brookes (1987a, 1987b) applied the stream power concept to investigate post-project readjustment of straightened river channels in some European countries. He found that straightened reaches in rivers with less than 15 to 25 W/m^2 of specific stream power at the bankfull stage tended to respond morphologically through processes led by deposition, while those in streams with powers in excess of 25 to 35 W/m^2 were likely to respond through erosion (Fig. 5.4). However, it should be noted that subsequent experience has shown that specific stream power cannot be used in isolation to infer either the potential for post-project adjustment, or the type of instability likely to occur.

In the case of catchment-scale sediment assessment, the purpose of screening is to allow prioritization of reaches for further investigation, with the limited resources being targeted on the riskier locations, and more advanced, quantitative investigations being reserved for those locations and situations that really merit such advanced treatment. The advantage of stream power analysis in this context is that it provides a fairly rapid and low-cost means of assessing sediment issues throughout an entire drainage network, using readily accessible variables (specifically, the reach-averaged discharge, width and slope, all at bankfull stage). However, while these are in essence simple variables to characterize, experience has shown that the accurate measurement of slope is crucial and that care must be exercised in selecting a slope value that reliably represents bankfull conditions in the study reach (Mant 2008).

Case example: River Caldew

The Caldew in Cumbria, UK, was the subject of Environment Agency investigation following severe flooding in Carlisle, in January 2005. Sediment issues arose because gravel accumulation within the urban reach of the river was identified as contributing significantly to flood risk. Operationally, deposition in a channel with a flood defence function would usually be managed by dredging, but as the river is designated as both a Special Area of Conservation (SAC) and a Site of Special Scientific Interest (SSSI) there is a presumption against sediment removal under the Habitats Directive. Dredging may still be allowed, due to the overriding public interest, but it is now a requirement to demonstrate that any work is performed in the most environmentally friendly way possible and to show that a need for repeated dredging continuing indefinitely into the future is avoided. Hence, sediment dynamics were investigated as part of a wider geomorphological assessment of 14 km of the River Caldew (Fig. 5.5).

It was concluded that the River Caldew is characterized by a series of dynamic reaches featuring bar growth (storing mainly coarse sediment) and localized bank erosion (yielding mainly fine sediment), interspersed with morphologically stable reaches that transfer both coarse and fine sediment downstream. This suggests that the dynamic reaches could be allowed to evolve naturally so that they store more of the coarse sediment (which is what causes problems in the urban reach) while releasing fine sediment that is not involved in sedimentation in the urban area. These findings provided the basis for design of an iSIS Sediment Model that established the potential for reducing the frequency with which future maintenance might be required in the urban reach by allowing or enhancing morphological evolution of the channel in the reaches naturally storing coarse sediment upstream (Wishart *et al.* 2007).

Limitations

Even as a screening tool, stream power analysis has marked limitations. Firstly, the tool should be

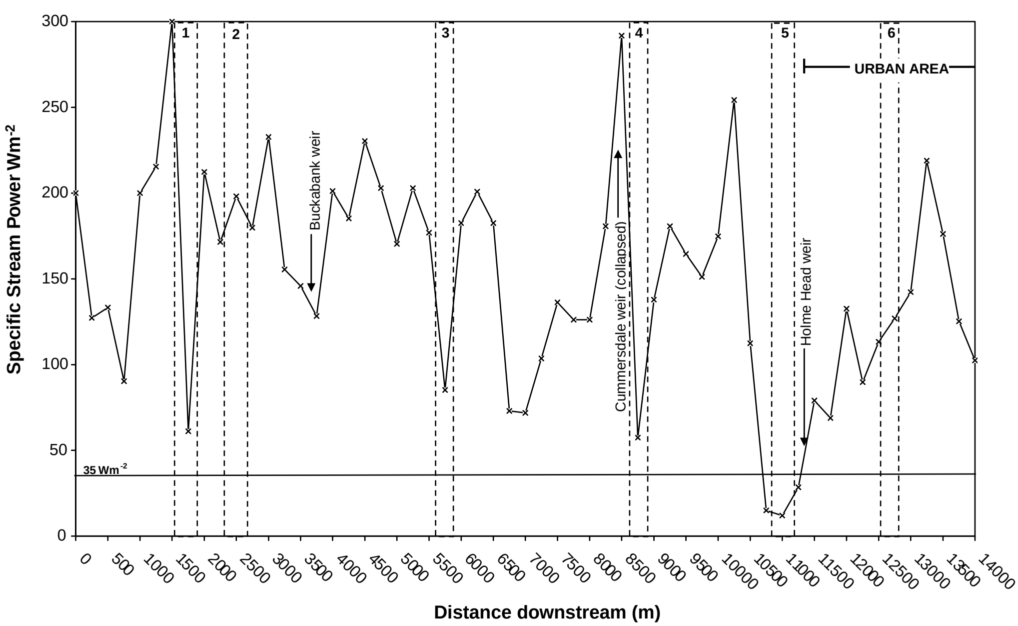

Fig. 5.5 Longstream variation in specific stream power in the River Caldew. Modified from Brookes and Wishart (2006).

applied only to alluvial (self-formed) channels with discernible floodplains. Secondly, it is best to apply stream power analysis at the reach scale so that the length of the channel unit used to calculate stream power is sufficient to avoid 'noise' associated with local features, but short enough that the spatial distribution of reaches captures the degree of complexity necessary to characterize the fluvial sediment transfer system (Worthy 2005).

Thirdly, it must be borne in mind that there are, in fact, no universal, threshold values of stream power that may be used to predict channel response to perturbation *a priori*. For example, Brookes (1988) reported a 'grey area' between about 25 and 35 W/m^2 within which rivers might respond to channelization through **either** erosion or deposition-led adjustments, and experience has shown that threshold bands vary significantly between rivers of different types located in different morphogenetic regions.

A fourth limitation is that the empirical datasets and indicative rules that currently underpin interpretation of observed stream power values reflect responses to channelization, adjustments occurring within channelized reaches (especially in straightened channels) and the impacts of river restoration in perennial streams. These data and the rules based on them cannot be used to predict adjustments in other environments or to other types of channel disturbance.

Finally, the results of a stream power analysis cannot be used in isolation to make reliable predictions of morphological responses to disturbance. At least a basic knowledge of the bed sediment characteristics and bank material properties is also required as these variables condition the morphological outcomes of process-response mechanisms and strongly influence the pattern and sequence of channel adjustments in perturbed streams (Simon and Thorne 1996).

In dealing with these limitations, Brookes and Wishart (2006) recommend that end users develop their own personal approaches to the accurate interpretation of the outcomes of stream power analysis and compile regional datasets appropriate to the river environments, boundary conditions and sediment issues in question.

River Energy Audit Scheme (REAS)

Background and development

A component of the research undertaken during Phase 1 of the FRMRC and led by Wallerstein and Soar (2006) centred on developing a catchment-scale approach for characterizing river reaches in terms of their potential to erode, deposit and transfer sediment, achieved by validating and building on the insights gained from qualitative assessment of sediment dynamics using a Fluvial Audit. The aim was to produce a practical and robust tool that could bridge the gap between simple approaches, such as the stream power screening tool, and complex sediment transport modelling in providing a scheme for identifying locations of potential instability within the fluvial system where more detailed analysis and resources could then be directed.

The theoretical basis for the REAS stems from the idea first proposed by Bagnold (1966) that excess specific stream power can be used to predict a stream's capacity to perform work on its channel boundaries through eroding and transporting sediment. However, recognizing the practical constraints and uncertainties associated with predicting sediment transport rates, REAS does not employ sediment transport calculations or attempt to route sediment through the river channel network. Instead, it calculates a measure of 'Annual Geomorphic Energy' (AGE) for each delineated reach by integrating excess stream power available for performing geomorphological work with flow duration (represented by a flow frequency histogram). Balances and imbalances in AGE between consecutive reaches are considered to be indicative of channel stability or potential for morphological change through processes dominated by scour (an excess in AGE) or those dominated by deposition of sediment (a deficit in AGE).

For each reach, the excess specific stream power for the median discharge in each discrete class in the flow frequency histogram is calculated and then multiplied by the water surface width to give excess total stream power per unit length of channel. These values are then multiplied by their respective discharge's decimal frequency of occurrence and finally summed for 'm' discharge classes to yield a time-integrated value of excess stream power, Ω_e:

$$\Omega_e = \sum_{j=1}^{m} F_j W_j \left[\sum_{i=1}^{n} P_i (\omega_j - \omega_{ci}) \right] \qquad (5.2)$$

where F_j = decimal frequency of occurrence of each discharge class, j, W_j refers to the channel width for each discharge class, ω_j = corresponding specific stream power (W/m^2) as defined in Equation 5.1, ω_{ci} = critical specific stream power (W/m^2) for initiating transport of the median grain size of each size class, i, in the particle size distribution of n classes, and P_i = decimal frequency of occurrence of each size class in the particle size distribution.

The critical specific stream power is estimated using Ferguson's (2005) expression, below, based on a reanalysis of Bagnold's original equation (1980):

$$\omega_{ci} = 0.113 D_b^{1.5} \log \left[\frac{0.73}{S} \left(\frac{D_i}{D_b} \right)^{0.4} \right] \left(\frac{D_i}{D_b} \right)^{0.6} \qquad (5.3)$$

where D_b = median particle size found in the bed material (mm); D_i = median particle size found in size class i (mm), and S = channel slope.

The result of Equation 5.2 is given as stream power with units of watts (per unit channel length) but can be expressed as an 'annualized' quantity of excess energy (for an average year in the period of flow record), or Annual Geomorphic Energy (AGE), in units of kilowatt-hours (kWh), where one kWh is the quantity of energy equivalent to a steady power of 1 kW running for 1 hour (or 3.6 megajoules of energy consumed). By making this adjustment, the final value represents energy consumption rather than rate and is comparable to the more conventional measure of annual sediment yield.

Further details of the development and a preliminary application of REAS are provided by Soar *et al.* (in press).

Data requirements and outputs

The input variables required for each reach in REAS (Fig. 5.6) are:

1 bed material particle size or size distribution;
2 flow duration curve;
3 channel cross-section;
4 channel slope;
5 channel and floodplain roughness coefficients (Manning n values).

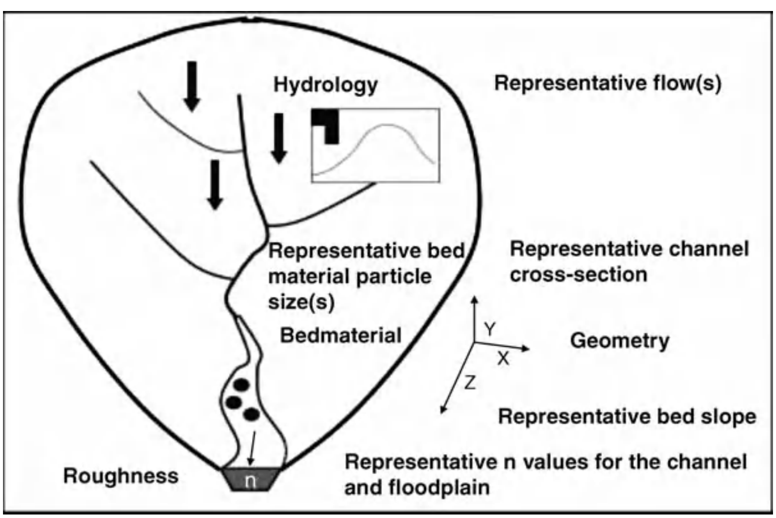

Fig. 5.6 Input data required for each sediment reach when using REAS (River Energy Audit Scheme) to analyse the spatial distribution of Annual Geomorphic Energy (AGE) in a river network.

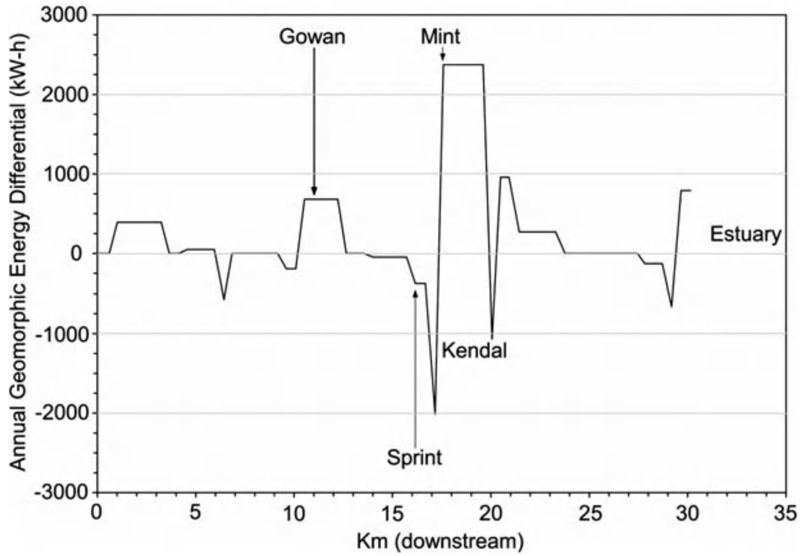

Fig. 5.7 Annual Geomorphic Energy (AGE) differentials along the River Kent, Cumbria, UK. Modified from Soar *et al.* (in press).

The values selected must be capable of representing conditions in the reach as a whole. Further details of data requirements and methods employed to ensure adequate data coverage are discussed by Soar *et al.* (in press).

The raw output of the accounting process is a series of balances, or differentials, in annual geomorphic energy (ΔAGE) between consecutive reaches (where reach number, r, increases in the downstream direction), according to the following expression:

$$\Delta AGE_{(r)} = AGE_{(r-1)} - AGE_{(r)} \qquad (5.4)$$

A positive value indicates that the subject reach has less annual geomorphic energy than the reach immediately upstream, which suggests a net deficit in sediment-transporting potential that might lead to disequilibrium led by sedimentation (i.e. a sediment sink reach). Conversely, a negative value indicates that the subject reach has an excess of annual geomorphic energy compared to that immediately upstream, which suggests a net gain in sediment-transporting potential that might lead to disequilibrium through net scouring of the channel (i.e. a sediment source reach). The results can be presented as a line graph illustrating how the AGE differentials change over the length of a watercourse, or they can incorporated into a GIS to identify reaches prone to geomorphic change and focus the resources available for more detailed sediment analyses. An example of the output from REAS, for the River Kent in Cumbria, UK, is illustrated in Figure 5.7.

While cross-sectional data are needed to perform the REAS computations, it does not mean that a detailed survey of the entire catchment is required as existing survey datasets can be supplemented by spot measurements (e.g. from River Habitat Surveys, or targeted field visits). In cases where data are particularly limited, cross-sections can be simulated from a digital elevation model with the channel component interpolated between spot measurements. One tool with this functionality is MAT (Modelling Assistant Tool), created by JBA Consulting for the Environment Agency in England and Wales, which, although subject to a degree of uncertainty, can provide acceptable and cost-effective solutions at the catchment scale.

Limitations and uncertainties

There are several sources of uncertainty in REAS, related to issues of data quality, processing and interpretation. Given the limited data availability in the UK and the perhaps unavoidable sparsity of hydrological, morphological and sediment data sampling sites when working at the catchment scale, uncertainty is inevitably associated with measurement and interpolation of channel dimensions, slopes and sediment sizes, the assignment of roughness values and the synthesis of flow duration curves, which often come from donor sites. However, with data availability dictated by (often quite limited) project resources, a degree of uncertainty is an inevitable outcome of this type of scheme and should not necessarily be treated as a limitation to its application.

Additional sources of uncertainty in the processing stage stem from assumptions made in the application of the various equations and the delineation of reaches of energy similarity (see Soar et al., in press), although the latter is optional as each cross-section in the scheme can be treated as its own reach, if this is preferred to reach averaging.

Finally, the scheme is based on interpretation and characterization of imbalances in annual geomorphic energy as being indicative of discontinuities in sediment transfer. However, this premise neglects the expenditure of energy in lateral erosion and the transport of material derived from bank erosion and sourced from outside the channel, which in some situations could constitute a significant proportion of the sediment load. In such cases REAS results might be discordant with reaches identified as sediment sources, transfers and sinks in a Fluvial Audit.

In light of these points, Soar *et al.* (in press) stress that results should be interpreted as being indicative, with a view to guiding sediment investigations towards reaches where limited resources are best targeted to develop a fuller understanding of sediment dynamics and the sensitivity of river morphology to disturbance. Hence, application of REAS should be seen as being complementary to the use of other tools in the FRMRC Sediment Toolbox, rather than as a stand alone alternative to any of them.

Sediment Impact Accounting Method embedded in Hydraulic Engineering Centre River Analysis System (HEC-RAS/SIAM)

Background and utility

The Sediment Impact Accounting Method (SIAM) is available in the 'Hydraulic Design' module of version 4.0 of the US Army Corps of Engineers' Hydrologic Engineering Center, River Analysis System (HEC-RAS) (Biedenharn *et al*. 2006; Gibson and Little 2006). It uses hydrological and hydraulic information computed using the one-dimensional, quasi-unsteady flow model in HEC-RAS to calculate the reach-averaged rate of bed material load transport by grain size, for the recorded range of discharges. Computed transport rates are integrated over the recorded range and durations of flow to compute an annualized bed material load transport capacity for each user-defined geomorphic reach. The capacity of the reach to transport bed material load is then compared to the annualized input of bed material load from upstream and local sediment sources (bank erosion, landslides, gullies, field erosion, etc.) to estimate the balance between bed material load supply and transport capacity in the reach for each size class. In SIAM, it is assumed that the movement of wash load is supply, rather than transport, limited and so no transport capacity is computed for wash load.

The Sediment Impact Accounting Method tracks the wash and bed material loads separately as the calculations progress downstream through the fluvial system. Each size fraction is classified as either wash or bed material load on a reach-by-reach basis, following convention usually attributed to Einstein (1950), that wash load is that part of the sediment load that is not found in significant quantities in the bed. In HEC-RAS/SIAM the threshold between wash and bed material loads is defined by the user for each sediment reach by selecting the appropriate bed material grain class. In practice, the grain size for which ten percent of the sample is finer (D_{10}) of the bed material is most

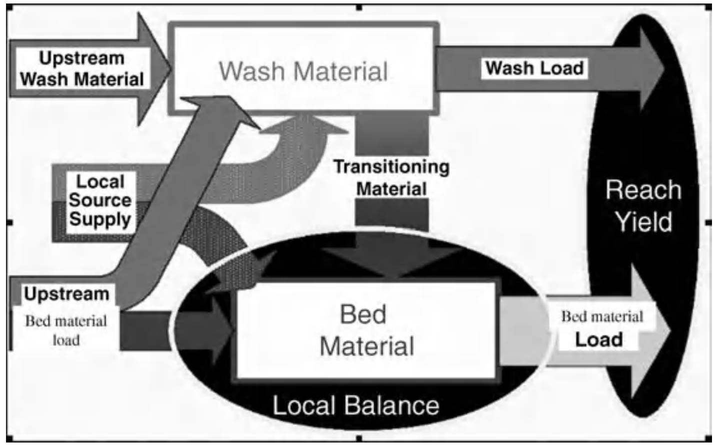

Fig. 5.8 Flow diagram illustrating how SIAM (Sediment Impact Analysis Method) accounts for bed material and wash load dynamics in the sediment transfer system. (See the colour version of this figure in Colour Plate section.)

commonly used for this purpose and, based on experience gained in numerous SIAM applications, it is a reasonable 'default' value for the wash load threshold. To apply HEC-RAS/SIAM it is therefore necessary to sample the bed in each sediment reach using a technique capable of accurately representing the finer limb of the particle size distribution. As the D_{10} changes downstream, sediment that is wash material in one reach may become bed material load in the next, and vice versa (Fig. 5.8) and SIAM tracks and accounts for these changes in calculating the reach-scale sediment budget.

Data requirements and outputs

The data required to run HEC-RAS/SIAM define for each sediment reach the:

• annualized hydrograph (discharge record and flow duration curve: to support calculation of quasi-steady, hydraulic parameters within HEC-RAS);

• channel form (geometry and roughness: to support calculation of quasi-steady, hydraulic parameters within HEC-RAS);

• bed material properties (particle size distribution: to support calculation of annualized bed material transport capacities by size fraction in HEC-RAS and to identify D_{10} (or another user-defined size fraction) for the wash load–bed material load threshold);

• sediment supply from local sources (average annual yields and particle size distributions: to support calculation of annualized sediment inputs in SIAM).

Local sediment sources include diffuse catchment erosion, landslides, eroding channel banks, tributaries, and anthropogenic sources such as arable fields, ditches, gullies and mines. Where sediment is removed from a reach for flood risk management or mineral extraction, the sediment impacts can be explored by including this in HEC-RAS/SIAM as a negative local source.

HEC-RAS/SIAM indicates the net sediment balance in a sediment reach. It does not, however, indicate the types or patterns of morphological change likely to result from any sediment imbalance. This is the case because it is not a sediment routing or mobile boundary model and there is no computational feedback between the sediment movement, flow hydraulics and channel form. Morphological interpretation of the results of a HEC-RAS/SIAM study requires additional information and the insights that may be obtained from other types of sediment-related investigation based on some form of stream reconnaissance, such as a Fluvial Audit or a Geomorphic Dynamics Assessment (Thorne *et al.* 2010).

Case example: Judy's Branch, Illinois

Judy's Branch is located near Glen Carbon, Illinois – across the Mississippi River from St Louis, Missouri. The goal of the HEC-RAS/SIAM application was to assist in identifying sediment management actions that would enhance habitat quality and reduce flood damages in a wetland by reducing the input of sediment to the wetland from Judy's Branch by 70%. Options to control downstream sediment delivery included excavating a large basin to trap sediment entering the wetland, or controlling the sediment sources upstream, and HEC-RAS/SIAM was used to investigate the feasibility of source control.

Measures proposed for sediment source control included: vegetative filter strips (VFS), local sediment basins (SB), and drop structures (DS). HEC-RAS/SIAM was used to assess the sediment impacts of these measures, based on 48 sediment reaches delimited by tributary junctions and points of significant change in stream hydrology. Figure 5.9 shows the sediment yield reductions predicted by HEC-RAS/SIAM for the various source control measures. The results indicated that source control was a feasible sediment management alternative to sediment trapping in a large basin, as a 70% reduction in sediment load could be achieved through implementation of all three measures together (VFS, SB and DS in Fig. 5.9).

Limitations

HEC-RAS/SIAM is a method of accounting for sediment that, in terms of model complexity and resource requirements, falls between the semi-qualitative evaluation generated by Stream Power Screening and the quantitative routing of sediment routing that is possible using HEC-RAS 4.0 or iSIS Sediment. However, its results lack the precision of a sediment routing model. When the risks associated with sediment dynamics are serious, but limited resources preclude sediment transport modelling, HEC-RAS/SIAM may add value to purely qualitative analysis and reduce uncertainty concerning sediment impacts to a level that is tolerable. It is particularly useful in fluvial systems where the characteristic behaviour of a significant fraction of the sediment load switches from that of wash load to that of bed material load within the problem or study reach.

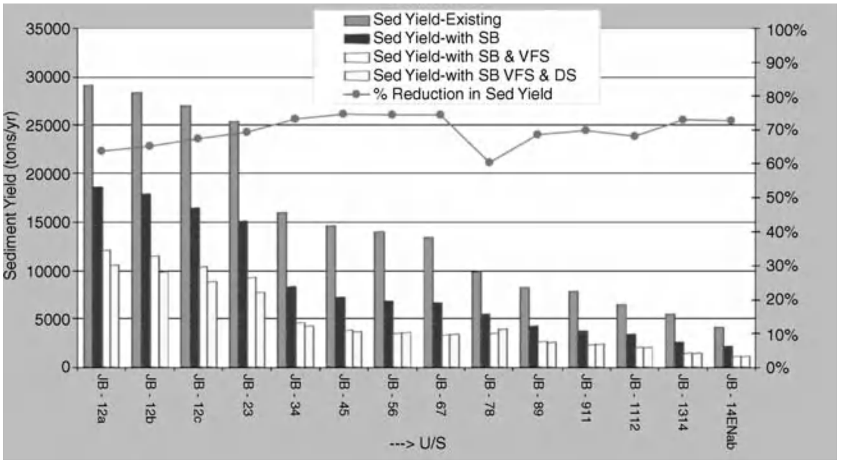

Fig. 5.9 Annualized sediment yields in Judy's Branch predicted by SIAM (Sediment Impact Analysis Method) for existing conditions and following implementation of a range of sediment source control measures. DS, drop structures; SB, small sediment basins; VFS, vegetative filter strips. (See the colour version of this figure in Colour Plate section.)

HEC-RAS/SIAM does not update the hydraulics or channel geometry to reflect any scour or deposition predicted in a sediment reach, meaning that its results indicate only the short-term trend of morphological change due to a sediment imbalance. An outcome of this is a tendency to somewhat over-represent erosion and deposition compared to values actually observed in the field. Also, as a reach-based model, HEC-RAS/SIAM uses reach-averaged parameters and produces reach-averaged results; it yields no information on the distribution of erosion or siltation within a reach. Consequently, the impacts of local scour or deposition are not simulated.

Finally, in its current form, HEC-RAS/SIAM implicitly assumes that the channel is alluvial: that is, it is free to adjust to scour driven by an excess of bed material transport capacity compared to the supply from upstream and local sources. Consequently, users must identify any non-alluvial sediment reaches, where scour is prohibited by naturally erosion-resistant materials or artificial stabilization encountered in the bed or banks. This may be achieved based on stream reconnaissance (Thorne 1998) coupled with bed material sampling to identify areas of erosion-resistant substrate and locate artificial bed and bank protection structures.

Hydraulic Engineering Centre, River Analysis System (HEC-RAS) Version 4.0

Development and basis

In 1976, the US Army Corps of Engineers developed the mobile boundary model HEC-6 (US Army Corps of Engineers 1993; Thomas 1994). This DOS program has remained an industry standard throughout the USA even while other popular HEC hydrological and hydraulic models (e.g. HEC-1, HEC-2 and UNET - unsteady flow through an open channel **Net**work) have been superseded by more powerful and user-friendly products (e.g. HEC-HMS and HEC-RAS). Recently, however, most of the capabilities available in HEC-6 have been incorporated into Hydrologic Engineering Center's open-channel hydrodynamic model, the River Analysis System (HEC-RAS), making use of the robust, existing hydrodynamic capabilities of RAS and providing helpful user interfaces for one-dimensional modelling of sediment transport.

The initial version with a sediment routing capability (HEC-RAS 4.0) uses one-dimensional, cross-section-averaged hydraulic parameters obtained from RAS's hydraulic engine to compute sediment transport rates and update the channel geometry based on sediment continuity calculations. The hydraulic computations are explicitly coupled with calculations of sediment transport, erosion, deposition, bed mixing and cross-sectional change. The result is a continuous simulation of cross-sectional change as sedimentation processes respond to the inflowing water and sediment hydrographs.

Computational methods

Hydrodynamics

Flow specification for sediment transport computations currently follows the 'quasi-unsteady' flow approach used in HEC-6. An event or period of record is approximated by computing a series of steady flow profiles. HEC-RAS uses each steady flow profile to develop transport parameters for each cross-section. Durations are assigned to each profile to define the temporal extent of the associated hydrodynamics and to route sediment movement. Usually, however, bathymetry updates are required more frequently than the flow increment duration, so a computational time step must also be specified. Channel geometry and a new steady flow profile are computed at the beginning of each computational time step, even if the flow remains unchanged (Fig. 5.10).

Transport calculations and cross-section updating

The seven transport functions currently available in HEC-RAS are: Ackers and White (1973), Engelund and Hansen (1967), Laursen (1958), Meyer-Peter and Muller (1948), Toffaleti (1968),

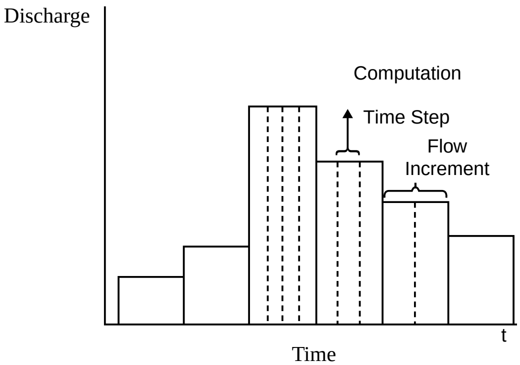

Fig. 5.10 Schematic to illustrate quasi-steady flow division used in characterizing hydraulics for sediment transport calculations in HEC-RAS 4.0 (Hydrologic Engineering Center River Analysis System 4.0).

Wilcock and Crowe (2003) and Yang (1972). HEC-RAS simulates graded sediment transport by dividing the sediment gradation curve into up to 20 discrete, editable size classes. HEC-RAS calculates an overall transport capacity by computing independent transport potentials for each size class. The sediment transport function is applied to each grain class as if it were the only material in the channel (with the exception of the Wilcock equation, which includes hiding functions and other inter-grain class dependencies). The transport capacity is computed for each grain class by multiplying the computed potential for that grain class by the relative fraction of that grain class in the active layer of the bed.

Bed elevation rises and falls in response to a sediment supply deficit or surplus in the control volume, i.e. the positive or negative difference between capacity and supply. HEC-RAS solves the Exner equation separately for each grain size – adding material to, or removing it from, the active layer. At the end of each computational time step, aggradation or degradation is translated into a uniform bed change over the entire wetted perimeter of the cross-section. HEC-RAS updates elevation information for each cross-section and performs new hydraulic computations before computing transport capacity for the next sediment routing iteration.

Physical constraints to erosion and deposition

HEC-RAS applies temporal erosion and deposition modifiers as well as sorting and armouring routines to augment the simple continuity computations. Physical process constraints are necessary because simply solving the Exner equation translates 100% of computed sediment surplus or deficit immediately into deposition or erosion. This does not reflect the fact that both deposition and erosion take time. Therefore, HEC-RAS applies time-dependent modifiers to the surplus or deficit calculated for each cross-section.

Deposition efficiency is calculated by grain size, based on the fall velocity and the expected centre of mass of the material in the water column – based roughly on Toffaletti's depth-concentration relationships (Vanoni 1975). A similar relationship is implemented to limit erosion temporally. HEC-RAS uses a 'characteristic length' approach adapted from HEC-6, which includes the assumption that erosion takes a distance of approximately 30 times the depth to develop fully.

Sorting and armouring

The other major process considered in the computation of continuity is potential supply limitation as a result of bed sorting. Currently, HEC-RAS has two options to compute the effects of bed sorting processes: Exner-5, a 'three layer' algorithm taken from HEC-6 (Fig. 5.11), and a simple 'two-layer' active layer method. Exner-5 divides the active layer into two sublayers, simulating bed coarsening by removing fines initially from a thin surface layer. During each time step, the composition of this surface layer is evaluated and if, according to an empirical relationship, the bed is partially or fully armoured, the amount of material available to satisfy excess capacity can be limited.

The simplified, 'two layer', active bed approach – with the Toro-Escobar *et al.* (1996) exchange increment method – is designed for simulating

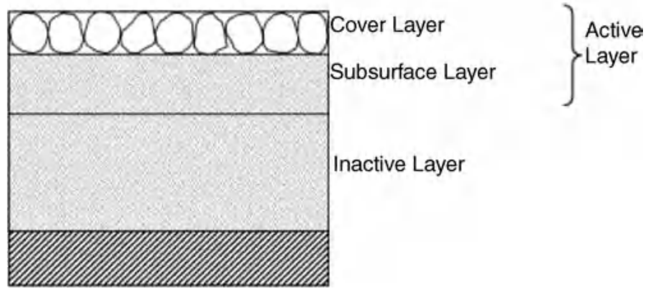

Fig. 5.11 Schematic of the way the bed is represented using the Exner 5 bed sediment sorting and armouring method in HEC-RAS 4.0 (Hydrologic Engineering Center River Analysis System 4.0).

gravel transport. Active layer thickness is set to the D_{90} of the bed material.

Limitations

The model has generally performed well in testing against HEC-6 and flume data, but its outcomes can differ slightly from those of HEC-6 under some circumstances due to minor differences in hydraulic computations. HEC-RAS includes a convenient user interface to specify the necessary data for a sediment analysis and a wide range of available outputs for troubleshooting and interpreting a simulation.

Limitations that should be considered when applying HEC-RAS include:

1 HEC-RAS is subject to the fundamental limitations of any one-dimensional sediment model. Dimensional issues are exacerbated in sediment analysis, as transport processes in nature are laterally heterogeneous while application of a one-dimensional model to a river sediment study involves lateral averaging.

2 HEC-RAS only accounts for vertical adjustments of the bed, so applications involving lateral migration and/or bank instability may be poorly characterized.

3 While quasi-unsteady flow approximations are often appropriate, careful thought should be given to the implications of representing a natural hydrograph with a series of backwater profiles. This may be an inappropriate approximation for applications to steep or 'flashy' systems and rivers with significant flow reversals or large reservoirs.

4 It is important to note that sediment modelling results are extremely sensitive to the transport equation selected and that the results for different equations can differ by an order of magnitude or more. Therefore, it is important to carefully select and calibrate the sediment transport function when developing a mobile bed model.

5 As HEC-RAS currently routes sediment by solution of the continuity equation rather than an advective or advective-diffusive (ADE) scheme that would tie transport to water velocity, careful consideration should be given to cross-section spacing and sediment movement to ensure that sediment transport speed does not significantly exceed water velocity (and preferably that it approximates actual sediment velocity).

6 One of the implications of the dimensional and non-ADE solutions in HEC-RAS is that it is most appropriately applied to sediments in the sand and gravel size ranges. While rough algorithms are employed within HEC-RAS to represent the transport, erosion and deposition of fine, cohesive sediments, additional care is necessary when attempting to apply HEC-RAS to channels formed in cohesive sediments and with complex flows, such as estuaries.

Despite these limitations, decades of successful applications have demonstrated that the one-dimensional approach employed in HEC-RAS can be useful when analysing a range of sediment-related problems and particularly those involving non-cohesive sediments in fluvial systems. However, users must be cognizant of the limitations listed above, and apply the model within this framework.

HEC-RAS is public domain software and is internationally available for download from the HEC website (http://www.hec.usace.army.mil/).

iSIS Sediment and Issues in One-Dimensional Sediment Modelling

Background

One-dimensional (1-D), computational models of river flow have been used since the 1960s (e.g. SOGREAH 1963) and are firmly established as the standard technique for routine analysis of river hydraulics and hydrodynamics. In the UK, a significant proportion of all main rivers have had at least part of their length modelled using commercial 1-D software packages such as iSIS, HEC-RAS or Mike11, and the current Environment Agency flood mapping budget of around £10 m per year contributes further to the stock of available models. This represents a significant investment in survey, calibration and assembly of one-dimensional models and a valuable resource not only of information but also of people experienced in use of these models.

The hydraulic parameters needed for a sediment transport calculation are common to those used in the hydraulic model calculations and thus the existing model packages are suited for adaptation to incorporate a sediment transport analysis. Commonly used European codes such as iSIS, Mike 11 and SOBEK have modules to compute sediment fluxes and bed level adjustment at cross-sections, and the same is true of programs developed in the USA, including HEC-6, GSTARS, Fluvial 12 and, most recently, HEC-RAS 4.0. It can, thus, be argued that the one-dimensional approach has utility because it builds on existing information and models and so is a logical first option when a fully quantitative analysis of sediment dynamics is required.

There are, however, practical and theoretical difficulties in the application of 1-D models that are not addressed in the relevant user manuals. These mainly relate to data input and the steps involved in running the model. As highlighted by Hayter (2002), experience shows that while it is relatively easy to create a 1-D sediment model using any of the available programs, accurate interpretation of sediment modelling results requires substantial knowledge and insight into the processes and mechanics of sediment transport and transfer in the fluvial system in question. While this is understood by a small corps of experienced sediment modellers, it does not always seem to be fully comprehended by the wider community of flood risk modellers and stakeholders.

Recognizing this, some of the major issues that complicate 1-D modelling of sediment dynamics are discussed here for the benefit of those new to, or on the fringes of, sediment modelling and the users of sediment model outputs in the context of flood risk management. As iSIS is currently the hydrodynamic model most widely used in the UK, it and its sediment module provide a suitable vehicle for considering these issues. However, the points raised herein are equally applicable to most of the available sediment transport programs and modules.

Development of iSIS Sediment

iSIS Sediment originated in 1994 as part of the iSIS collaborative venture between Halcrow and HR Wallingford. The earlier code developed at HR Wallingford is described by Bettess and White (1981). iSIS Sediment was initially developed for simulation of siltation in large irrigation canals such as those in Pakistan, but with the addition during the 1990s of routines for graded sediments and improved accounting for layering effects, its capabilities and applications widened rapidly. In 2001, further extensions were developed and tested at Herriot Watt and Glasgow Universities (Schvidchenko *et al.* 2001), although the research version has not, to date, been completely incorporated into the standard code.

A licence for the iSIS hydrodynamic model is required to run the model though there is currently no additional charge for access to the sediment module.

The model can be used in a number of ways, including long-term simulations of bed evolution or to explore the effectiveness of engineering interventions such dredging or modification of the channel to improve flood conveyance or deal with a sediment-related problem.

Features of the model include:
- use of the iSIS interface for model preparation, simulation and display of results;
- coupled simulation with a fully hydrodynamic model;
- user may specify different bed compositions at every cross-section and a different gradation for the inflow of sediment;
- looped, branched and tidal channels may be modelled.

Limitations include:
- the need to edit the sediment file in a text format;
- sediment transport calculations are based on composite cross-sectional properties;
- a restricted set of iSIS units can be used excluding, for example, interpolated sections and fixed roughness formulation based on Manning's n.

Conceptual basis

It is not necessary to recount the detailed basis for iSIS Sediment as the computational framework used in the hydraulic and sediment transport calculations is well documented in the user manual (iSIS 1999). The sediment simulation is based on calculation of sediment transport rates and an accounting of erosion and deposition using the concept of 'layers' of sediment with a 'well mixed' distribution of sediment sizes. The difference between the quantities of sediment arriving at and leaving a computational cross-section is simply represented as a mass balance calculation based on the Exner equation:

$$(1-\lambda)W\frac{\delta z}{\partial t}+\frac{\partial G}{\partial x}=0 \qquad (5.5)$$

where λ = bed porosity; W = water surface width (m); z = bed elevation (m); t = time (s); G = sediment transport rate (m^3/s); and x = distance in flow direction (m).

The bed layer concept (Fig. 5.12) is used to account for sorting between active and parent layers that is especially important in gravel-bed rivers, where there is a wide grading of sediment, and surface layers may be significantly coarser than the parent material. Bank material is not specified separately, and so is implicitly assumed to be of the same composition as the bed material, which is seldom the case in an alluvial stream, especially in gravel and cobble-bed rivers.

In the standard version of iSIS the sediment transport equations available are Engelund and

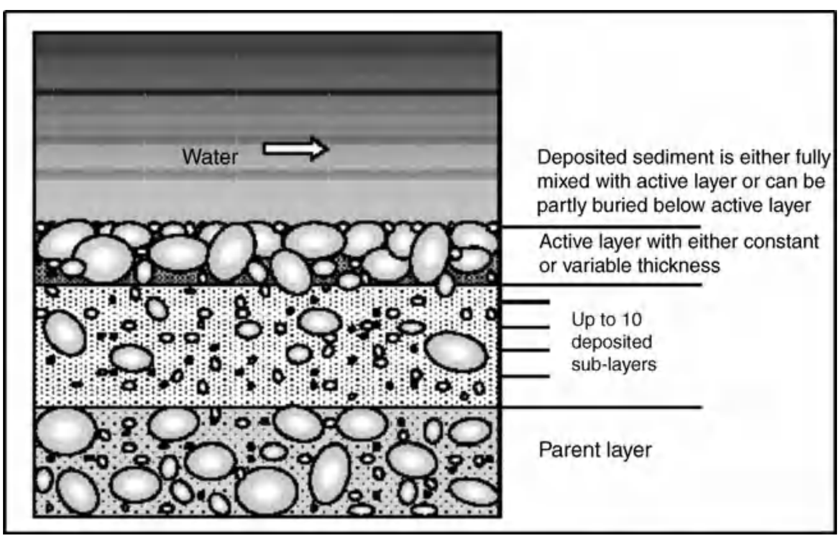

Fig. 5.12 Bed Layer Concept used in iSIS (2001).

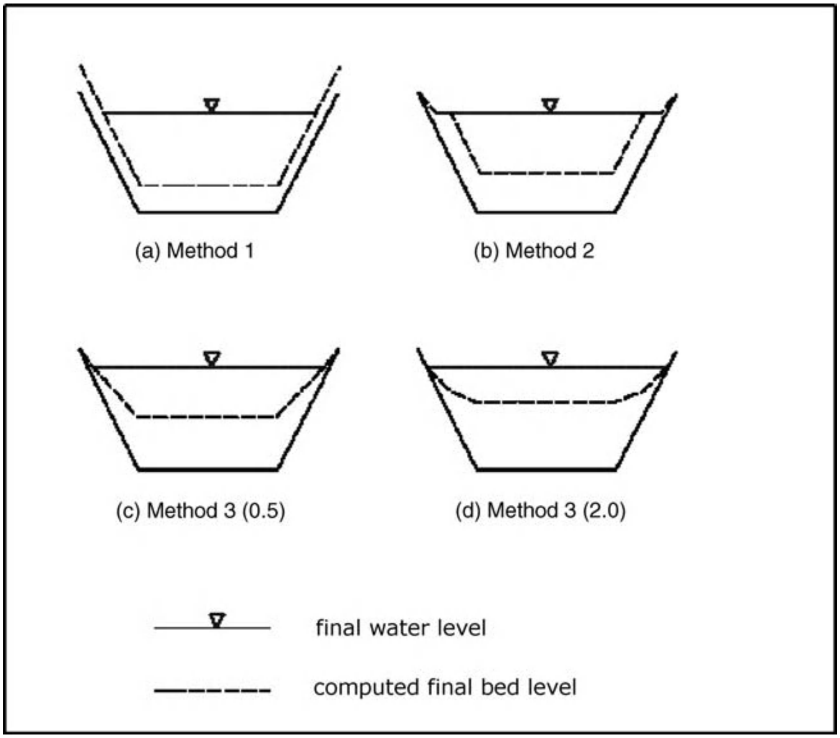

Fig. 5.13 Bed Updating Options in iSIS (2001). (a) Method 1 spreads erosion or deposition evenly across the whole cross-section. (b) Method 2 distributes erosion or deposition across only the wetted cross-section. Method 3 distributes erosion or deposition across the cross-section according to a user-specified exponent on the local flow depth. Examples show distribution scaled on (c) the square root of depth and (d) depth squared.

Hanson (1967), Ackers and White (1973), Ackers and White - as updated by Ackers (1993) and, for fine, silts Westrick and Jurashek (1985). Other sediment transport formulae are available in the research version of the code (iSIS 2001). The sediment transport calculation utilizes the cross-sectionally averaged hydraulic properties supplied by the hydraulic computation. This may cause complications where there are divided channels or floodplain flows. Also, the flow intensity specified for the threshold of sediment motion has a significant influence on the performance of the model.

A number of options are available in iSIS Sediment to handle how erosion or deposition are laterally distributed across cross-sections when the bed elevation is updated (Fig. 5.13) and these have major implications for model predictions.

Issues in 1-D Sediment Modelling

There are a number of key issues that need to be addressed when planning any sediment investigation that involves 1-D modelling. Many of these issues are particularly important in iSIS Sediment because it takes greater account of fluvial processes and the characteristics of natural sediments than many other programs. Issues include:

1 Selecting limits to the modelled reach – this is necessary because it is seldom feasible or justifiable to model the entire fluvial system.

2 Selecting the period and duration to be modelled and deciding how to represent the flow – ideally, continuous simulation is preferred, but for long model runs it may be necessary to keep run times manageable by modelling only the sediment transporting events.
3 Identifying the cross-section spacing needed to represent sediment processes adequately – cross-sections must be spaced more closely in a sediment model than is permissible when modelling the flow of water alone.
4 Setting the sediment in flow to the modelled reach – often there will be few or no measured data and yet this boundary condition is important to how the model will run. Usually, if no data are available, the sediment input is set equal to the transport capacity at the first computational section, so that this boundary is stable.
5 Deciding how to handle sediment grading – a decision must be made on whether it is necessary to model sediment transport by size fraction and, if it is, the range and number of size classes to be used, how many samples must be collected to characterize the size distribution sufficiently well, and how much complexity to use in the model (this ranges from using a different size distribution for every cross-section to using a single size distribution to represent the bed throughout the modelled reach).
6 Selecting the sediment transport equation – based on the scale and steepness of the watercourse, the sediment size distribution and the model complexity, it is essential to select a sediment transport equation appropriate to the application.
7 Identifying the simplifying changes necessary to an existing hydrodynamic model required for it to support the sediment module – for example, successful simulation of in-bank sediment transport may require reducing extended cross-sections or using spills to remove flood flows to out-of-bank storage.
8 Working out how best to represent in-stream structures that impact sediment dynamics – sediment traps, bridges and culverts and out-of-bank flows around these structures disrupt connectivity in the sediment transfer system and this must be reflected in the behaviour of the model. It is widely recognized that adequately representing the sediment impacts of hydraulic structures is a challenge in practically all sediment models.
9 Optimizing run parameters such as choice of layer thickness, critical flow for sediment entrainment and method of cross-section updating – how these user-defined parameters are set can have a major impact on model outcomes.

Experienced modellers recognize many of these issues and the pitfalls that lurk behind them and they are adept at dealing with them too. They can do this because they have first-hand knowledge of how to address the issues and avoid the pitfalls, based on how simulations have been finessed in the past despite software limitations and weaknesses in process knowledge, such as how in-channel sediment transport is affected when flow spills over bank. Unfortunately, the guidance available in the literature on dealing with these issues is currently limited and improved dissemination of both research findings and experience gained during practical applications could help improve the reliability and accuracy of 1-D sediment models that are built and run by less experienced modellers. For example, 20 years ago, Samuels (1990) provided clear guidance on the minimum spacing necessary to properly represent flood hydrodynamics in a 1-D model, but up to now no similar guidance has been published for sediment modelling.

Limitations to 1-D Sediment Modelling

Having dealt with the issues underlying 1-D sediment modelling, there remain limits to its applicability that cannot be worked around easily or simply. These include:
1 Three-dimensional effects of sediment transport such as local scour at bridges are not accounted for.
2 Sediment calculations for compound channels and overbank flows may be less accurate due to the lateral averaging of flow parameters.
3 Layers of sediment are assumed to be homogeneous across the width so that lateral sorting, such as occurs at meander bends, is not represented.

4 A fixed Manning's n is used and this is applied to both channel resistance and bed shear stress used in sediment transport calculations.
5 The effect of bedforms (dunes, pebble clusters) on resistance and sediment mixing is not included explicitly.
6 Simulation of gravel traps that are small relative to cross-section spacing can be difficult.
7 Steep rivers with high Froude numbers may affect model stability and breach the applicability of the available sediment transport equations.
8 Accurate simulation of armouring effects is dependent on the criteria selected for deciding on active-layer thickness, which are poorly defined.
9 Some of the more widely preferred sediment transport formulations for gravel-bed rivers, which have been incorporated into the research version of iSIS, are unavailable in the standard program.
10 Bank erosion/instability, its sediment yield and its morphological impacts are not well represented in the standard model.
11 Selection of the method for updating cross-section geometry is subjective, but can strongly influence the modelling results.
12 Reliable interpretation of results depends to a degree on user experience.

The conclusion that must be drawn is that uncertainties associated with 1-D sediment modelling are high and that this uncertainty is derived from lack of knowledge concerning sediment transport mechanics and the sparse availability of field measurements of bed sediment size distributions and sediment loads in UK rivers, as well as limitations in the performance of 1-D models themselves. As uncertainties in multi-dimensional models are also conditioned by limited knowledge of processes and data with which to characterize sediment properties and dynamics, they are likely to be as great as or greater than those in 1-D modelling. This suggests that the use of simple, fast-running 1-D sediment models within stochastic or probabilistic frameworks may at present be the best way to handle uncertainty when predicting future sediment dynamics.

The fact remains though that, as Bradley *et al.* (1998) conclude, provided that reasonable calibration and verification data are available, and that an experienced modeller undertakes the work, available 1-D sediment models like iSIS Sediment remain good predictive sediment tools for application to flood risk management.

Cellular Automaton Evolutionary Slope and River Model (CAESAR)

Background and basis

The Cellular Automaton Evolutionary Slope and River Model (CAESAR) is a two-dimensional flow and sediment transport model that can simulate morphological changes at the catchment or reach scales, on a flood by flood basis, over periods up to several thousands of years. To date, CAESAR has been applied to over 20 different catchments and reaches, at spatial scales ranging from that of a 500 m reach to that of a 500 km^2 catchment, and over timescales ranging from that of an individual flood to 10,000 years.

CAESAR is a cellular model that may be classed as a 'reduced complexity' model. It fits in the gap between multi-dimensional hydrodynamic and Computational Fluid Dynamics (CFD) models that can readily be applied to sediment-related analyses of small reaches over short timescales, and coarse-resolution landscape evolution models that can simulate changes in regional landforms over thousands of years. Cellular landscape models work by representing the terrain using a grid of cells, within which landscape development is determined by fluxes of water and sediment between the cells that are simulated using rules based on simplifications of the governing physics (Nicholas 2005). In fluvial geomorphology, cellular models use simplified or 'relaxed' versions of the complex flow equations used in CFD models. This allows a substantial increase in speed of operation, which, in turn, enables them to be applied to extensive reaches and large catchments over long timescales.

CAESAR was inspired by a model of river braiding reported by Murray and Paola (1994). Based on some of Murray and Paola's approaches,

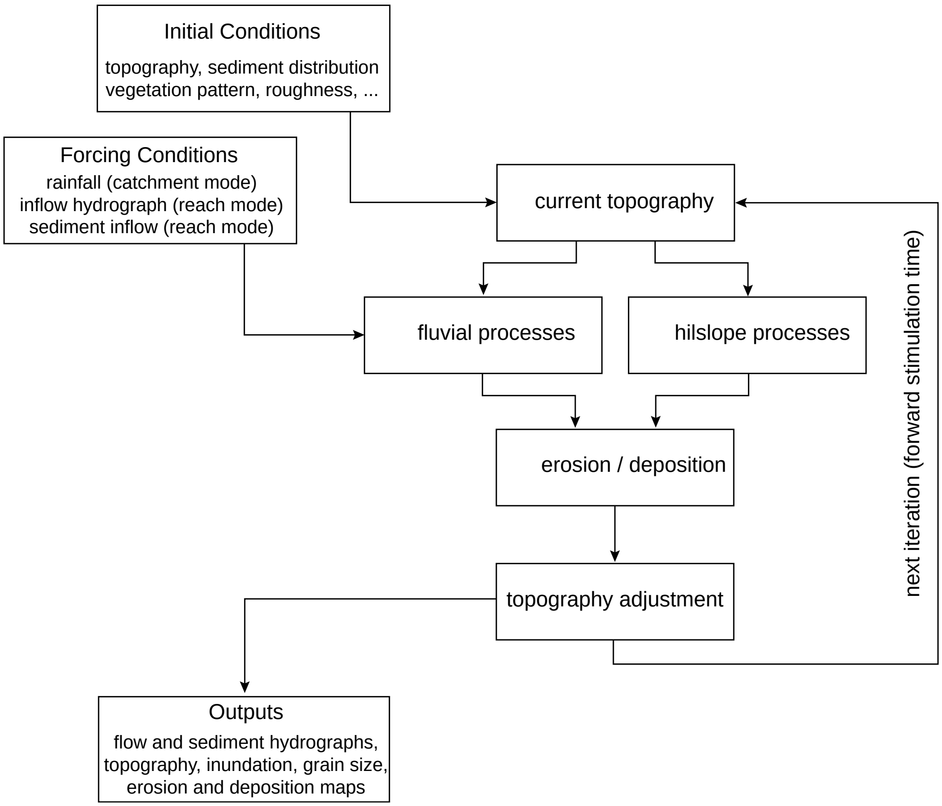

Fig. 5.14 Flow diagram to illustrate operation of the Cellular Automaton Evolutionary Slope and River Model (CAESAR).

a cellular automaton model of river catchment evolution was developed and this subsequently led to the CAESAR model. Detailed descriptions of the model and its development have previously been published (Coulthard *et al*. 2000, 2002, 2005) and are not reproduced here. However, Figure 5.14 summarizes how the model functions schematically.

CAESAR is coded in Visual C#, and runs as a Windows program on Windows NT, 2000 and XP. No programming experience is required to use it. Example files can be downloaded and the program run within minutes. Applying it in practice requires the capability to manipulate and edit Digital Elevation Model (DEM) files, and users will require some basic knowledge about data manipulation using (e.g.) Microsoft Excel. The source code for CAESAR is openly available for download under the terms of a GNU – General Public License (http://www.gnu.org/), which prevents it from being sold for profit.

Data requirements

CAESAR can be run in two modes: the catchment mode, with no external fluxes or inputs aside from rainfall; and the reach mode, with one or more points where water and sediment are inputted to the system.

For the catchment mode, CAESAR requires hourly rainfall data. Ideally, the study catchment should have such a rainfall record as well as a gauged point or outlet so that the hydrological model can be calibrated. However, if this is not available nearby rainfall data can be used, and there are ranges of example settings from which the hydrological model can be parameterized.

CAESAR requires a raster DEM (not a Triangular Irregular Network - TIN) of the catchment, and editing and correcting the DEM is an important part of preparing for a CAESAR simulation. The model has been applied with DEMs having grid cell sizes ranging from 1 m to 100 m. The choice of grid cell size is important, as this allows compromises to be made between the area that can be modelled, the spatial resolution, and the run time. CAESAR can run with up to 2 million grid cells, but is best suited to arrays with 250,000 to 500,000 cells. DEMs often contain errors that can cause the model significant problems, and it is therefore recommended that DEMs are preprocessed to remove any pits that act as internal sinks, and to ensure that the drainage network follows a straightforward descent to the exit point, which must be located at the bottom righthand corner of the DEM. This can be carried out simply using the freely available ARC-HYDRO extensions toolkit for ARC-GIS 8.x and 9.

In reach mode, rainfall data are not required but it is instead necessary to specify the water and sediment inputs for the reach.

CAESAR also requires information on the characteristic grain size distributions in the catchment. The model can accept up to nine different grain size fractions and account for sediment moving by both bedload and suspended load transport mechanisms.

Limitations and uncertainty

CAESAR is an experimental tool for scientific research and hypothesis testing. It is based on physical processes but uses approximated flow and sediment transport rules to replace the complex governing equations of fluid flow and sediment transport. The accuracy of these approximations, and how they are affected by their application in a 2-D model, are largely unknown. Uncertainty in the veracity of CAESAR's computational engine is compounded by difficulties in validating the results of a CAESAR run. Some outcomes can be relatively easily validated. The flow model, for example, can be compared against measured flood outlines or inundation maps generated by 1-D hydrodynamic models such as HEC-RAS or iSIS. Conversely, suitable data to validate model predictions of erosion, deposition and channel planform evolution are more difficult to find.

A limitation of fluvial models in general, which is certainly applicable to CAESAR, is that the heterogeneity of natural environments presents a major problem. For example, spatial and temporal variability in bed roughness, climate-related fluctuations in catchment runoff and sediment yield, and changes to vegetation and land use all influence the behaviour of a river system. Our inability to predict runoff and sediment responses to future anthropogenic impacts introduces uncertainties that are particularly difficult, if not impossible, to represent in the input parameters for the model, or indeed to replicate deterministically within the numerical simulation.

A further limitation common to all sediment models is the considerable uncertainty that surrounds the sediment transport rules used to drive the model. In CAESAR they are far from ideal, and no generally applicable equation from which reliable rules can be derived has yet been developed – for further discussion see Coulthard *et al.* (in press). At present it would, therefore, be unwise to put too much faith in the absolute sediment fluxes generated by CAESAR. Rather, the value of the model lies in its use for revealing relative changes, for example when seeking to establish whether increases in flood frequency or magnitude are likely to cause more erosion or more deposition in a project reach. Indeed, experience of using CAESAR has shown that its greatest strength is in simulating system-scale patterns of erosion and deposition. For example, it can consistently differentiate between those river reaches that are more likely to be eroding/incising and those that are probably depositing/laterally unstable.

Finally, the ethos of CAESAR's development is very much one of openness. The code is open source, model applications are freely available and limited support is offered via a discussion board that can be found on the CAESAR website. This means that the development of the code can be somewhat *ad hoc* and sporadic, and that the level of user support provided cannot match that available for a commercial package. However, this approach does mean that CAESAR is freely available for download from Tom Coulthard's website (http://www.coulthard.org.uk).

Closure

The tools outlined here span a range of requirements in terms of the data, technical expertise and resources (time and money) necessary to support their application; they generate output resolutions that range from indicative to diagnostic; and they can be applied at spatial scales from river reaches to whole catchments. They are, consequently, suitable for addressing a wide range of sediment-related issues in flood risk management.

None of the tools is perfect; indeed, all are subject to potentially serious errors through misapplication, and uncertainties in their outcomes remain large. In describing them, the authors have been careful not to exaggerate their capabilities and to be candid in reporting their limitations. They have done this because individuals intending to perform sediment assessment, analysis or modelling-related investigations and those who are the end users of their outputs must understand that broad-scale sediment dynamics are difficult to characterize and even more challenging to quantify or predict. So why attempt this? Because quite simply, the weight of evidence suggests that future sediment-related flood risks are far too important to be ignored (Lane and Thorne 2007, 2008).

Selection of a method or model appropriate to the task at hand is a crucial first step in any sediment analysis. In this context, it is vital that users weigh the need to perform the work in a timely and cost-efficient manner against the nature, extent and severity of potential sediment-related risks, so that uncertainty regarding the management of future sediment dynamics and associated flood risks may be reduced to a level that is acceptable or, at least tolerable.

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6 A Measured Step Towards Performance-Based Visual Inspection of Flood Defence Assets

GAVIN LONG AND MICHAEL J. MAWDESLEY

Background

In the UK the Environment Agency (EA) is tasked with the management of a large flood defence infrastructure of approximately 70,000 assets with a total length of over 35,000 km (Halcrow, 2004). These assets are periodically inspected by qualified staff and assigned a condition grading based on their visual condition. The current method of inspection is effective for assessing the visual condition of an asset but is thought to produce an inadequate assessment in terms of likely asset performance under loading.

Asset monitoring and inspection is a primary input to any asset management system. Without assessments of condition and likely performance of the assets within an infrastructure system it is impossible to provide effective decision-making regarding the management of such systems. Accuracy and recency of inspection data are key determinants in the success of any asset management system.

Visual inspection is the simplest form of non-destructive testing as it does not require any specialist equipment. It is a widely used technique for monitoring assets. It must be noted that any visual inspection is prone to inaccuracy as it can only assess the surface details of an asset; changes to interior structure and condition are not easily identifiable until they lead to changes to the surface of the asset. Visual inspection of flood defence assets in most of the UK has been carried out by the Environment Agency (EA) since its inception in 1996.

Aims and Objectives

The work described here was undertaken under the auspices of the Flood Risk Management Research Consortium (FRMRC) and funded by the Engineering and Physical Sciences Research Council (EPSRC), the Department of Food and Rural Affairs (Defra) and the EA. Its primary aim was to produce a revised methodology for the visual inspection of flood defence assets, to the proof of concept level, which represented an advance towards performance-based assessment.

A more detailed list of objectives is:

1 To achieve a measured step forward in the visual inspection of flood defence assets.
2 To introduce a scientific methodology into the process wherever possible.
3 To highlight areas where the scientific methodology is lacking.
4 To provide a vision of how the inspection should be done in the future.

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This chapter is intended to provide only a basic overview of the method proposed and the step forwards it represents from the existing method of visual inspection. The full report (Long *et al.* 2006) provides a far more detailed description of the revised method of visual inspection, including guidance materials for assessing all major asset types. Objectives 3 and 4 in the list above are not covered in this chapter due to space restrictions but are discussed in the full project report.

Existing Method of Visual Inspection for Asset Condition

In the UK, flood defence assets are regularly inspected (the regularity of inspection is determined by the criticality of the defence) in accordance with EA guidelines and procedures and by trained inspection staff. Currently, this requires the inspector to walk the length of the asset examining its cross-sectional elements, such as the crest, faces or slopes. The inspector assigns a 1 to 5 grade ('Excellent' to 'Very Poor') to each cross-sectional element of an individual asset. The grading of an element relies on the inspector's training and through reference to the Condition Assessment Manual (Environment Agency 2007). This manual along with the training provided to inspectors should help to ensure consistency of assessment between different inspectors. The manual gives a textual description for each type of asset element and condition grade along with a representative photograph. The inspector's experience will also be an important factor in determining the assessment. If it is not possible for the inspector to access the element for visual inspection, an X is recorded under its condition to signify that the element condition was not able to be inspected.

The Condition Assessment Manual has been updated a number of times since the visual inspection procedures were instigated to reflect current research in terms of asset condition and performance. The most recent version of the manual (Environment Agency 2007) referred to here includes a number of changes that were made in light of findings of this and related work within the subject domain.

Asset cross-sectional elements are identified and coded via a simple diagram and table on the asset inspection form (Fig. 6.1). The inspection process starts with the inspector checking the coding of asset elements on the form and cross-section diagram showing the location of each element within the asset against the actual asset. Elements are coded according to type and material. In most instances the form will already be completed from the contents of the National Flood and Coastal Defence Database (NFCDD).

Figure 6.1 shows a completed inspection form for an embankment. Each element of the asset has been graded at condition grade 2 indicating good condition. The overall condition grade for the whole asset is often recorded as being the condition grade of the element in the worst condition (highest value). Simple formulae may also be applied to integrate the element condition values into an overall asset condition value.

In reviewing the current method of visual asset inspection it is important to focus on its primary aims and objectives, namely:

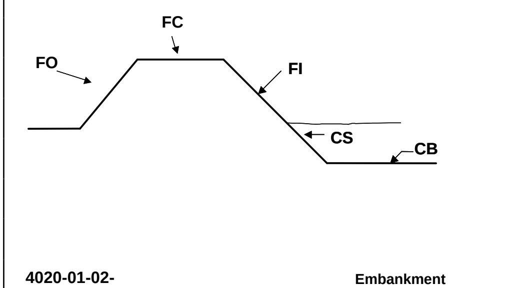
- To record a snapshot of asset condition.
- To provide accurate and repeatable assessments of asset condition.
- To be a practical solution balancing assessment accuracy against practicality. The visual inspection method should provide a quick assessment of condition without requiring expert (e.g. qualified engineer) knowledge.
- To produce an assessment of condition for all types of defence assets.
- To identify weak assets prior to their failure.
- To act as a decision support tool for asset management.
- To aid in the prioritization of maintenance options or new capital projects.

The current method fulfils all these objectives to some degree. However, it has weaknesses in many areas; in particular it does not provide a realistic assessment of likely performance of an asset. A list of specific weaknesses of the method is as follows:

Environment Agency : Flood Defence Asset Survey Form

								Surveyed By				Date	
Area	004	Sub Area	17	Watercourse	4020	Reach	01	Sub reach	02	L R or B	L	Tidal or NT	T
Defence Code	02		River Idle : Flood Bank with landward SSP wall)							Outfalls Only		No of pipes	
Structure Code										Height		Shape	
DS NGR L02	478958 394810			US NGR	478895 394881					Width		Length	
Length	90			Height of Defence	3.5	Major / Minor		Major					

Type	Sub Type	Mat	Revet	Cond	Slope	Width
CB	R	E				
CS	R	E	T	2	32	
FI	B	E	T	2	32	
FC	B	E	T	2	0	3
FO	B	E	T	2	27	



Recommendations

Fig. 6.1 A completed asset inspection form with condition grades assigned.

• Assesses condition of individual defence elements only. Asset performance may be linked to the condition of a number of elements or to specific features within key elements. The grading of elements is a practical solution to the visual inspection process but has no scientific basis in the determination of overall asset condition.

• There is no direct link under the current method between the visual condition and likely performance and reliability of the asset. In some instances, the current guidance produces condition grading at odds with the likely performance of an asset.

• The contribution of individual elements to overall condition under the current system is treated simplistically or not at all. There is some use of a criticality score for elements but this is not implemented consistently across regions of the Environment Agency and is not used to produce an overall condition grade for the asset itself.

• Overall condition grades for assets, where used, are poorly defined and based on the weakest element or an average of the elements. The inspection itself produces a list of condition grades for individual elements of an asset only.

• The guidance provided for condition grading of elements is too generic in many instances and covers a broad range of features. For example, condition grade 3 given to an embankment slope could indicate poor quality of grass cover, minor cracking or slipping of slope, or the presence of vermin infestation.

• The low range of potential condition grades produces high variability in assessments where the state of the element is a borderline value.

In addressing the weaknesses of the current method it must be emphasized that these are often intrinsically linked to its positive features and great care must be taken in the development of a revised method. This could easily produce a method that whilst eliminating the limitations of the current method also removes the benefits provided by it.

A Performance-Based Methodology for Visual Inspection

Some basic understanding of failure modes and performance modelling of defence assets is necessary to describe the method of visual inspection being proposed. Defence assets can fail in several ways. There are a number of well-known processes for failure, some of which have been modelled using physical or statistical models. These performance models can be used to determine the likelihood of a specific failure mode occurring given a set of data relating to the flood defence system being analysed.

There are three methods for devising performance models for an asset:

• **Historical data** – A statistical approach can be used to analyse historical records, where available, of asset failure. By comparing hydraulic loading conditions and any failures that occur, rough measures of likely performance can be produced. The advantage of this approach is that it does not require any detailed understanding of the physical processes involved in asset performance. However, it is highly dependent upon the quality and amount of historical data available for an asset.

• **Expert opinion** – This could be defined as being ill-recorded historical data. Experts familiar with the asset and local conditions could provide simple heuristics defining likely performance versus hydraulic loading. Expert opinion is widely used in early definitions of fragility curves for asset types under the Risk Assessment of flood and coastal defence for Strategic Planning (RASP) project (HR Wallingford & University of Bristol, 2004) and the condition indexing system employed by the United States Army Corps of Engineers (USACE) (McKay *et al.* 1999). This method enables performance to be assessed where there are few or no recorded data. It also takes account of local conditions. It is highly dependent upon the knowledge of the experts being used to assess performance and therefore lacks repeatability.

• **Scientific models** – These are generally based on analysis and synthesis of physical attributes of the asset backed up by experiment. They enable a calculation of performance to be made based on equations comparing loading with resistance. A scientific model, by definition, must be repeatable and therefore may provide a more reliable measure of performance than a model based on limited historical data or expert opinion. Problems with the use of scientific models of performance are often in obtaining accurate measurements of the model parameters for an asset.

Drawing upon current research into the performance modelling of defence assets (e.g. HR Wallingford 2003), a set of relevant failure modes and associated performance models was identified for use with the new methodology being proposed. The proposed method is described below and is referred to as the **condition indexingprocess**. The description starts with the key elements that form the basis for the method.

Performance features (PF)

Performance features are the building blocks on which the condition indexing process is based. They represent the elements of an asset that are to be inspected in order to produce the condition index. Performance features must possess a number of attributes. These are listed below in order of importance:

1 They must be related to at least one failure mode, and therefore to the performance of the asset – For a performance-based visual inspection, the primary objective is to inspect visual indicators of likely asset performance. If the visual condition of a feature plays no role in asset performance, it should not be inspected.

2 Visible – The feature to be inspected must be easily assessed on a visual inspection. For example, many of the performance models have geotechnical parameters that cannot be observed on a visual inspection. Their assessment requires there being a visible feature on the asset surface that directly or indirectly relates to their current value.

3 Gradable – In addition to being visually identifiable, the current condition of the feature must be able to be graded visually. There must be sufficient visual indictors related to the PF in order to assign the range of condition values associated with that PF.
4 Mutually exclusive – Ideally, PFs should be mutually exclusive. There should be no chance of mistaking the condition of one PF for another. However, if necessary, this attribute can be relaxed to some degree in order to satisfy the more important attributes 1, 2 and 3.

Performance features can apply to a single element of the asset and therefore be repeated for each element where relevant or can apply to the whole asset.

A PF could relate to a number of failure modes or be associated with a single failure mode. Performance features that are uniquely associated with a single failure mode are important in the differentiation of most likely failure mode and usually will have a high contribution rating linking them to that failure mode. An example of this is the class of PFs relating to deformation of the asset structure or cross-section.

The choice of performance feature is partly determined by what is possible to assess visually and also by the failure modes that need to be identified. Performance features must be chosen that will identify these failure modes directly or indirectly. Consistency of the method wherever possible is another factor considered in the identification of appropriate performance features. It is necessary to produce a set of performance features that balance accuracy of inspection with implementational issues such as the workload associated with inspecting an individual asset. Too large a set of performance features, though possibly producing a more accurate condition index, would increase the duration of an inspection beyond reasonable levels.

Condition rating

A condition rating is a numerical value representing the current condition of a performance feature determined by visual inspection. The value range of the condition rating should allow for the inspector to discriminate condition to an appropriate level of detail between the boundary conditions: excellent (high performance) and very poor (very low or zero performance). It was determined that the value range for the condition indexing method should match the existing condition values used (1–5) to ease any transition process in implementing a new method of visual inspection.

A slight change in the definitions associated with each of the 1–5 grades was needed to reflect the move from a method based purely on visual condition to one utilizing visual condition to assess likely performance. The entire 1–5 range of condition applies to all performance features inspected by the revised visual inspection method.

Contribution

Contributions represent the relative influence of a performance feature on a failure mode. They are the crucial link between performance and visual inspection of condition. Contributions are predefined values obtained from sensitivity analysis of performance models or, in the absence of performance models, from collective expert judgment. Contributions are used to calculate the failure mode indices for an asset as described later.

Any value scale could have been chosen to represent the contribution providing it gave enough differentiation of value to express the relative influence of all performance features or failure modes. For the sake of clarity and ease of use it was decided to use a contribution value range of 0–1 where the total of all contributions identified for each failure mode was equal to one. This ensures that all failure modes are weighted by the same total amount of contribution. It also simplifies the calculation of the failure mode indices as explained later.

The main obstacle in the use of contributions is obtaining accurate contribution values that reflect actual performance. The lack of accurate and extensive performance modelling data led to the use of expert opinion to obtain contribution value sets for the development of the measured step forwards proposed by this project.

Confidence

There is widespread use of confidence/uncertainty assessment in inspection methodologies, as it provides feedback about the precision of the inspection. Confidence, in terms of the method for visual inspection proposed, refers to the inspector's assessment of the accuracy of the condition value given to a performance feature. The confidence will be affected by:

- the experience of the inspector on the specific type of asset;
- the access that the inspector has to the asset;
- the prevailing local weather conditions at the time of inspection;
- tidal conditions at the time of inspection (where appropriate).

In the majority of instances the likely uncertainty range should be low and be spread above or below the assigned condition score. There may be instances where the uncertainty is high, for example when the performance feature is difficult to inspect due to poor visibility.

A number of methods that could be applied to assign confidence values to the condition score for a performance feature were considered. The chosen method involves the inspector assigning a signed value of confidence to a performance feature. This represents the likely difference between the assigned condition and actual condition. A positive score indicates that the actual value could lie in the range between the assigned value and the assigned value plus the confidence value. A negative score indicates that it lies between the assigned condition score and the condition score minus the confidence value. Unsigned values would indicate that the range of potential values lies in the range of the assigned condition index plus or minus the confidence value. For example: condition = 2, confidence = + 1, potential condition range = 2–3.

In those instances where the performance feature is not inspectable, the confidence score should not be used to reflect this lack of knowledge. Confidence assessment could be used to assign a condition of 3 with a confidence of ± 2 to reflect that the PF's condition could be anywhere within the 1–5 condition value range. The problem with assigning such a low measure of confidence is that the condition index produced using this method would have a very large uncertainty associated with it, and the 3 assigned to condition would skew the Failure mode likelihood Indices (FIs) and Confidence Indices (CIs). It was deemed a better option to grade non-inspectable PFs with a zero for their condition representing that no condition was assigned for those features. The calculation of FI and CI would exclude the zero-rated PF.

Failure mode likelihood index (FI)

The FI is a score in the same 1–5 range as the condition scores assigned to each performance feature. It is related to a specific failure mode for an asset type and is calculated using the formula below:

$$FI(Failure\ Mode) = \sum_{i=Performance\ Feature} Contribution(i) * Condition(i) / \sum_{i=Performance\ Feature} Contribution(i) \quad (6.1)$$

The nature of the visual inspection process can be highly uncertain for a variety of reasons such as poor weather conditions or restrictive access to assets when inspecting. This will inevitably lead to instances where certain performance features cannot be inspected with any degree of confidence. Any such incomplete inspection will therefore have a set of contributions totalling less than one for one or more failure modes. In this case the calculated failure mode indices will reflect this by the division by total contributions assessed. The fact that the FI was calculated on a reduced contribution total would be noted and may trigger the need for a re-inspection of the asset to check the omitted performance features. This could require visiting the asset at low tide or some other method for gaining access to the inaccessible features.

The confidence assigned to the individual PFs is used to calculate the minimum and maximum values for each FI. This produces the range of potential values for each FI. The larger this range, the greater the degree of uncertainty associated with that FI value. The position of the assigned condition FI within the value range calculated using the confidence values gives an indication of the nature of the uncertainty, i.e. whether it is more likely for the condition to be better or worse than the condition assigned by the inspector.

Condition index (CI)

The condition index for an asset represents some indication of its overall condition based on the assessment of performance features and the calculation of failure mode indices associated with those performance features.

The calculation of the CI could be achieved by the same method used to calculate the individual FIs. This would require a set of contribution values that represented the relative likelihood of each failure mode occurring for an asset. The problem with this approach is that there is no definitive dataset that could be used for the contributions at the asset-type level. The potential use of failure mode contributions needs further research to establish whether a set of appropriate values could be determined that would be useful in terms of visual inspection of asset performance.

An alternative method of calculating the asset CI is referred to as the **weakest FI** approach. This involves the CI being determined as the maximum value in the set of FIs. If the FIs for an asset with four failure modes were 2.1, 3.2, 2.2 and 3.7, the Asset CI is equal to 3.7. The problem with this approach is that it seems to ignore the FIs other than the one with the maximum value. However, it is proposed that in terms of measuring performance at the asset level, this is precisely what should be done.

Guidance and training

Guidance and training are essential to any method of visual inspection. They are the key components in ensuring the accuracy and consistency of inspections across a wide range of inspection staff potentially working within different operational environments. The nature of the changes being proposed as a measured step towards performance-based inspection is such that effective training and guidance will be critical to the success of the proposed method.

A number of mechanisms for providing guidance were investigated as a part of this project. Brief descriptions of these are listed below:

- **Condition grade descriptions and images** – as used in the current method of visual inspection for UK flood defences (Environment Agency, 2007). These provide a simple method for assessment of condition but are essentially limited due to their reliance on a single photograph. It was decided that this approach would be insufficient as the sole form of guidance for the condition indexing process.
- **Flowcharts** – provide a highly structured mechanism for assigning condition. The increased structure and fixed pathways in the flowchart ensure greater consistency of assessment than a textual description, which suffers from the ambiguous nature of natural language. Flowcharts display the process of condition assessment across all condition grades within a single chart. This allows the inspector to understand the differences between the grades of condition more easily. The production of flowcharts is a more complex process than a text description and must be extensively trialled to identify any errors and/or omissions. The rigid structure of the flowchart can also be a limitation as it will be impossible to include every possible situation and combination of factors that may be found on site. Training would need to emphasize the need to use the flowcharts as guidance material and not as a replacement for the inspector's knowledge.
- **Checklists** – the use of checklists is a common approach to visual inspection in many industries. The USACE Condition Indexing system (McKay *et al.* 1999) commonly employs checklists in the recording of asset condition. This method of guidance is a good way to ensure consistency of inspection but can require the assessment and

recording of a large amount of data therefore increasing the duration of the inspection process. It also requires that the checklists used are fairly extensive and detailed in their content. There is a danger that any findings from the inspection that do not fit into the checklist may be ignored.

The guidance method chosen for use in the condition indexing process was primarily flowchart based. This was felt to provide the structure needed to ensure consistency of inspection and, in addition is a simple-to-use method that does not require too much additional input from the inspector, unlike checklists.

Flowcharts were designed using some general principles. They start with a process box at the top of the chart describing the performance feature being assessed (and the failure mode it applies to where relevant). The five condition grades are listed along the bottom of the chart going from 1 to 5 from left to right. All flowchart decision boxes are answerable with a 'yes/no' or 'none/minor/severe' type response where possible. In those instances where this is not possible, the potential responses are explained where they are applied or within the notes attached to a chart.

The definitions of 'minor' and 'severe' referred to commonly in the charts refer to the following general descriptions:

- **Minor** – the item being assessed is only visible to detailed inspection of the performance feature. For example, a minor misalignment of the crest of a vertical wall would only be visible if the inspector were to closely examine the crest looking along it and comparing it with other wall sections.
- **Severe** – the item being assessed would be easily visible under a cursory inspection.

Using the previous example for the crest of a vertical wall, a severe misalignment would be obvious to the inspector as he or she approached the asset. It would not require close inspection to identify, and would be visible to a non-expert or member of the public.

To reduce ambiguity, notes have been attached to those charts where it was felt that the decisions to be made could be unclear. As a final point regarding guidance, it must be noted that guidance is only intended to **guide** the inspector and give examples of the assignment of condition to PFs. There are likely to be instances where on-site findings do not match exactly with the flowcharts or tables as it is impossible to cover every possible permutation of condition that could occur in practice. Inspectors must use their judgment and effectively use the confidence assessment where the assessment of a PF is ambiguous.

The Condition Indexing Process

A number of activities form the condition indexing process, and these will now be described to give a better understanding of the overall process being proposed.

Pre-inspection: data gathering and planning

Prior to the actual inspection, an inspector should obtain all the relevant data relating to the sites to be inspected. This could include previous asset inspection records, local geography and asset locations, design records, local geotechnical information, asset topography and the standards of protection afforded by the assets (i.e. their performance specification). The exact data requirements will also be determined partially by experience and knowledge of the assets to be inspected. Previous inspection data should always be examined to provide a comparison with the inspection to be carried out, to observe any changes to the asset occurring over time, and to highlight any known points of weakness for more detailed inspection.

Assessment of the performance features

This step of the process represents the actual on-site inspection of the asset. The inspector performs a detailed observation of the asset and assigns condition and the associated confidence scores to each PF in turn. The method for achieving this is left to the inspector's discretion. For the set of PFs relating specifically to a single failure mode (visible deformations of cross-section or obvious structural deformations), the inspector must be careful to establish the specific PF or PFs occurring as there are some similarities between them. The guidance (flowcharts and tables) provided attempts to reduce this problem by highlighting any distinguishing features for each.

If a particular failure mode is self-evidently occurring or is known to dominate for a given asset, the inspector has the option to ignore this step and move straight onto step 2 taking the 'Override' option. This allows the inspector to override the condition indexing process where there is an obvious failure occurring. This eliminates wasted inspection time in the assessment of performance features that will not be needed to identify the asset CI.

As with the current inspection method, the inspector should also be encouraged to add recommendations or notes onto the inspection record. This could be additional detail regarding the condition or confidence assigned to a PF, potential actions or maintenance work that is required, or notes regarding the conditions under which the inspection was carried out.

Calculation of the failure mode indices

This stage involves the calculation of the FI scores for an asset. This can be achieved in two ways:

- **Override** – Where a failure mode is self-evidently dominating the inspector can assign a condition score directly for the FI thereby eliminating the unnecessary assessment of PFs for an obviously occurring failure of the asset.
- **Standard** – Calculation of the FIs from the assessed PFs using contributions scores and the formulae given earlier.

Override should only be used where the nature of the failure occurring is obvious. This could be due to the inspector's detailed knowledge of the asset, data obtained in the pre-inspection stage and confirmed on site, or the imminent and self-evident failure of an asset. For example, if the inspector arrived at a vertical wall structure to find the wall leaning into the river at an angle with obvious and severe movement of the structure having occurred, the inspector could ignore the assessment of individual PFs and assign a condition grade 5 to the relevant failure mode (in this case, overturning). When inspectors override assessment of PFs they should note the reasons for this in their inspection record.

The standard approach to the calculation of the FIs could be performed during or after inspection.

Calculation of the asset condition index

This can be done on site or after inspection and be easily automated as with the previous step. Once the asset CI has been calculated, the condition indexing process is complete for that asset. The asset management system will then be used to analyse the inspection results and determine an appropriate course of action. As with the current inspection method, the inspector can decide whether to add a recommendation regarding the nature or urgency of any intervention required for the asset such as remedial works or replacement.

Condition indexing for earth embankments

Earth embankments form the majority of linear flood defences in the UK, especially in rural or semi-rural areas. They can have a very long lifespan given good local soil conditions and regular maintenance to ensure performance. Embankments are commonly subject to settlement over their lifespan due to the consolidation of the underlying soils. Local geotechnical or hydrological processes can cause movement or deformation of the embankment, which, if left unchecked or uninvestigated, can lead to failure of the embankment through a number of mechanisms, some of which will be described.

Embankments are susceptible to erosion of their fill material without surface protection. All earth embankments employ some form of surface protection to alleviate this problem. In many instances grass cover is sufficient to protect the embankment from erosion. Where hydrological loading is more extreme, due to erosive currents or regular overtopping of the embankment crest, other forms of revetment are applied to an embankment to reduce erosion of the embankment material.

Morris (HR Wallingford 2003) provides a comprehensive examination of embankment performance with analysis of significant, historical embankment failures. HR Wallingford's Performance and Reliability study (HR Wallingford 2004) on flood defence assets draws upon this work, amongst others, and includes a number of performance models relating to embankments.

Failure modes

There are a number of potential failure mechanisms for earth embankments. Some cannot be uniquely identified using visual inspection as they require geotechnical, and destructive, testing. Examples of failure modes for earth embankments are given below. A list of failure modes for embankments and other asset types is provided in the full report (Long *et al.* 2006)

Slope instability

Slope instability covers a range of failures identified from a number of sources such as shallow slips, deep rotational failures or sliding of sections of the slope (Fig. 6.2). The exact nature and cause of instability are hard to assess through visual inspection. Due to this, these various geotechnical processes have been grouped into this failure mode for the purposes of condition indexing.

In terms of the visual assessment of slope instability there are a number of signs of its occurrence, such as cracking, fissuring, movement of slope sections, slipping of slope and slumping or heaving of slope sections.

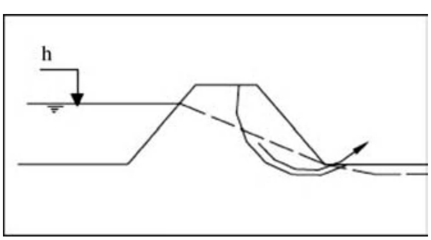


Fig. 6.2 An example of slope instability in an earth embankment (HR Wallingford 2004).

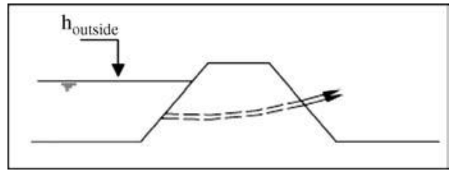


Fig. 6.3 Piping through an earth embankment (HR Wallingford 2004).

Piping

Piping can occur through or underneath an embankment structure (Fig. 6.3). Piping underneath an embankment is uncommon in the UK due to the nature of the soil types. Piping is caused by excessive seepage through an embankment leading to the washout of fill material. This can eventually lead to the formation of a pathway through or under the structure. Once this has been created, catastrophic failure can occur rapidly, often referred to as a **blowout**. Piping can be very hard to identify prior to imminent failure.

In terms of visual inspection for the occurrence of piping there are factors that should be assessed. The soil type making up the embankment is the dominant factor. Any asset or area prone to piping failures will need to be investigated closely.

Indicators of piping would range from actual springs of water appearing in the embankment (probably near the outer toe) to signs of saturation or pooling of water around the outer slope. The presence of animal burrows within an embankment could reduce the effective embankment width and provide an easy pathway for piping to occur.

Overtopping leading to breach

The overtopping of an embankment is not a failure in itself. The embankment is designed to hold back water up to the height of its crest and then be overtopped. However, if the overtopping is of sufficient force and duration it can lead to erosion of the crest and outer slope (Fig. 6.4). Significant erosion of the crest will reduce the standard of protection of the embankment and increase the likelihood of further overtopping. Significant

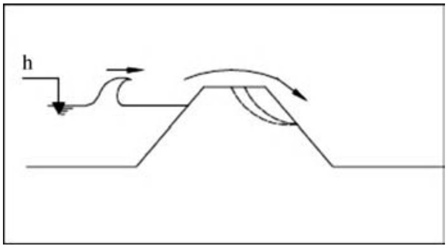


Fig. 6.4 Overtopping leading to breach failure mode (HR Wallingford 2004).

erosion of the outer slope will eventually lead to a breach of the embankment once the rear of the embankment has been sufficiently weakened.

Signs of this failure mode will be most evident in terms of erosion of the rear of the crest and the outer slope. The vegetation covering the outer slope is likely to be damaged and eventually lost due to the overflowing water. Grass cover acts as protection against erosion caused by overtopping, and the loss of this cover will be a key indicator that a damaging level of overtopping is occurring.

Performance features

A set of performance features were proposed in collaboration with industry experts based on research into embankment performance and failure modes, current UK visual inspection guidance and visual inspection methods used elsewhere. The 10 performance features identified are:
1 animal burrowing/vermin infestation;
2 foreign objects in the crest or rear slope concentrating the erosion process;
3 cracking and/or fissuring;
4 third-party damage (cattle, vehicles, etc.);
5 direct evidence of seepage or piping;
6 visible deformation of cross-section caused by piping;
7 visible deformation of cross-section caused by slope instability;
8 revetment condition;
9 erosion of cross-section;
10 vegetation condition (outer slope).

Contribution of performance features to failure modes

The contributions linking performance features to failure modes for an earth embankment are shown in Table 6.1. These were determined through a consensus of expert opinion combined with previous research work into the performance of embankments (HR Wallingford 2003), and were devised for use in the pilot trials of the condition indexing process.

Table 6.1 Contributions Linking Performance Features (PFs) to Failure Modes for an Earth Embankment

	Contributions				
Performance features	Slope instability	Revetment failure	Piping	Backfill washout	Overtopping leading to breach
Animal burrowing/vermin infestation	0.1	0.2	0.2	0.2	0
Foreign objects in the crest or rear slope concentrating the erosion process	0.1	0.2	0.1	0.2	0.2
Cracking and/or fissuring	0.3		0.1	0.2	
Third party damage (cattle, vehicles etc)	0	0.1	0	0	0.2
Direct evidence of seepage or piping	0	0	0.5	0.2	0
Visible deformation of cross-section	0.3	0	0.1	0	0
Revetment condition	0.2	0.5	0	0.2	0
Erosion of cross-section	0	0	0	0	0.4
Vegetation condition (outer slope)	0	0	0	0	0.2

Guidance for assessment of performance features

Some examples of the flowcharts used for guidance in assigning condition to the PFs are given in this section.

Figure 6.5 is the simplest of the set of flowcharts relating to embankments as it is an assessment of vegetation coverage. This is easily inspectable through a visual inspection. As this PF is only linked to the 'overtopping leading to breach' failure mode, the inspector is only required to check the vegetation coverage on the outer slope.

The flowchart shown in Figure 6.6 is used to assess the condition relating to animal burrowing

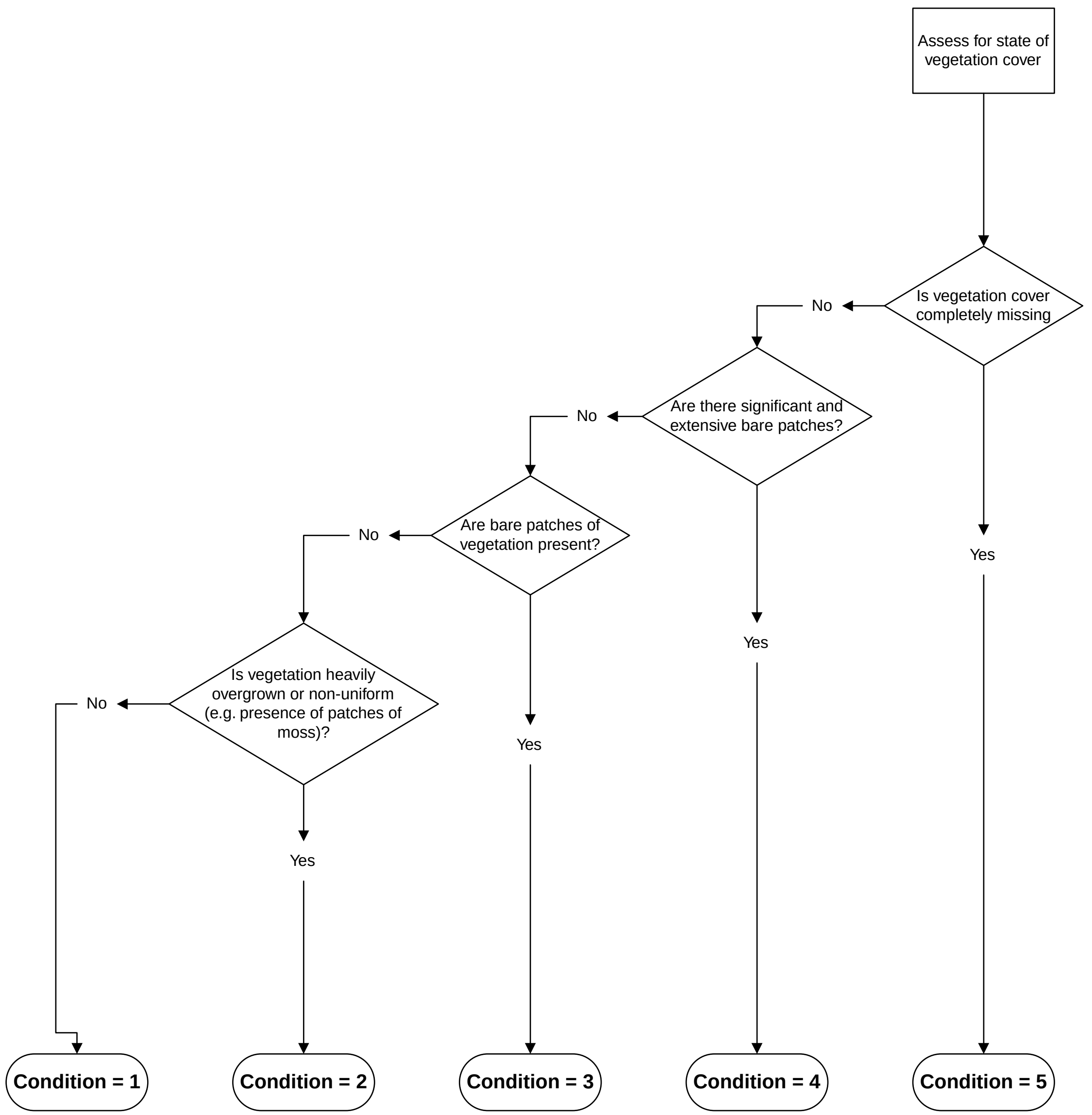


Fig. 6.5 Flowchart to assess the condition of vegetation coverage for an earth embankment.

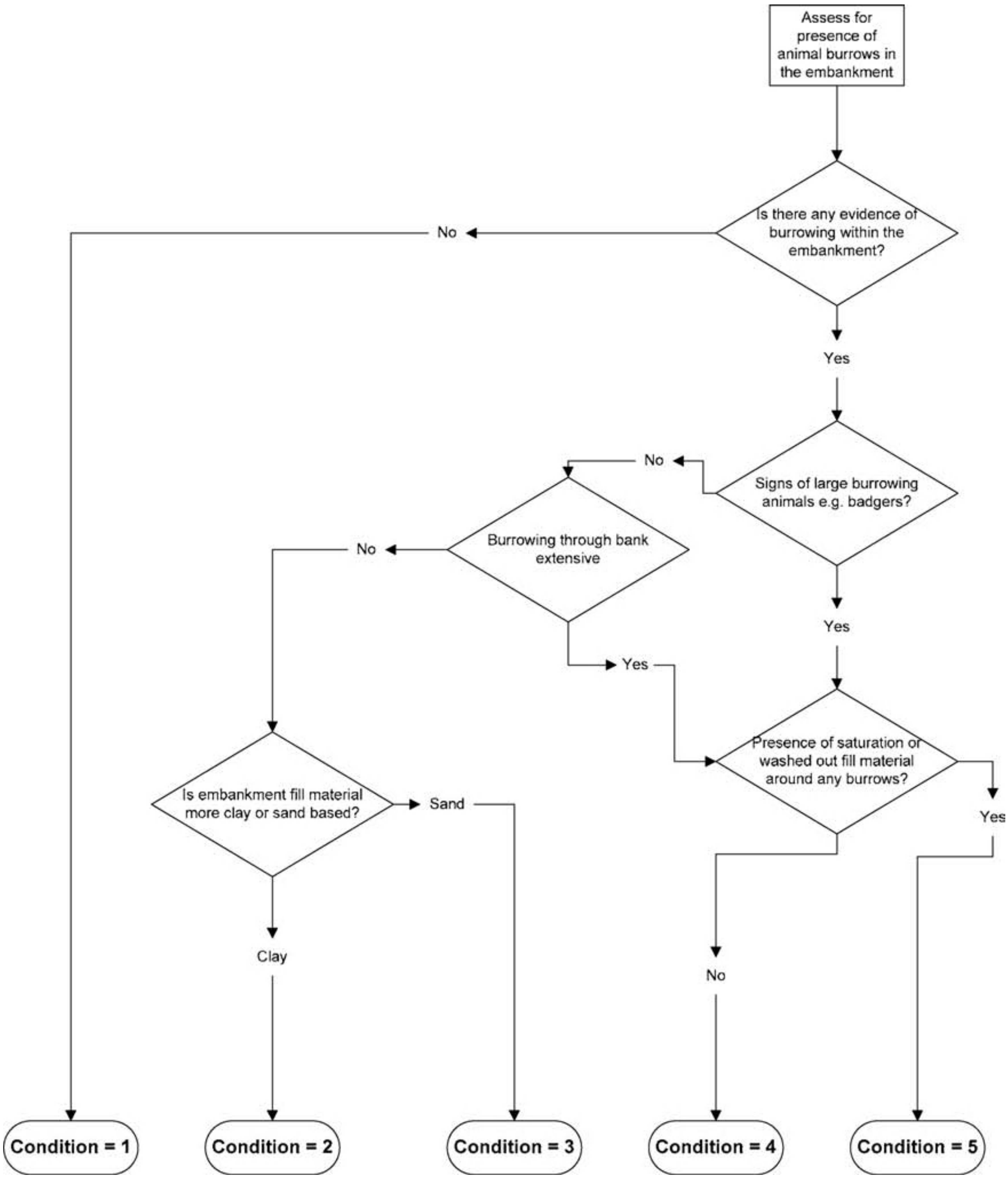


Fig. 6.6 Flowchart for the assessment of animal burrowing within an embankment.

in an embankment. Poor performance (grades 4 and 5) requires evidence of large burrowing animals or extensive burrowing. The very poor condition also requires the presence of washed out fill material to be evident. This indicates that seepage or piping is actually occurring. If burrowing is not extensive, the soil type of the fill material is used to assign condition grades 2 and 3.

Figure 6.7 assesses the embankment for any visible deformations of cross-section caused by slope instability. There are a number of forms of slope instability that can occur, and some are more serious than others in terms of performance. Cracking is a sign of slope instability but to ensure that there is no duplication with the 'cracking and/or fissuring' PF the chart initially

Fig. 6.7 Flowchart to assess any deformation of embankment cross-section caused by slope instability.

checks for any signs of slope movement or misalignment. Cracking without movement or misalignment is ignored under this PF and would be picked up under the 'cracking and/or fissuring' PF. Movement of the slope with the appearance of cracking parallel to the crest is a sign of instabilities deep within the embankment and is graded more severely than where cracking is not evident.

Case Study

To demonstrate proof of concept for the work, a number of case studies were selected. These were principally based on assets used to train potential inspection staff for the Environment Agency. Images are provided by John Chatterton Associates and the Middlesex University Flood Risk Hazard Research Centre. One of these case studies is now described to illustrate the condition indexing method.

The case study presented is an earth embankment that forms a section of the Shardlow ring bank in Derbyshire. The nature of the ring bank means that the asset is only subject to any hydraulic load under extreme conditions (at times of high water levels). This enables the visual inspection process to be carried out easily with both slopes and toes of the embankment visible under normal conditions. Due to its infrequent use as a flood defence and its position some distance from the watercourse, the asset is subject to some unusual third-party activity. Livestock grazing on the field in front of the inner slope have used the flood defence for access to the field leading to some damage to crest and slope (Figs 6.9 and 6.10). Also, a drain has been placed in the inner slope (as seen in Figs 6.8 and 6.11).

Step 1: assessment of performance features

The assessment of performance features used in the Case Study is shown in Table 6.2. The embankment is in relatively poor condition because of the damage caused by third parties and foreign objects as shown in the images. Vegetation on

Fig. 6.8 Embankment length and view along crest.

the outer slope is overgrown and has therefore been classified as grade 2. PF 7 has been given a confidence of -1 to indicate the assigned condition of 4 (poor) is uncertain and could actually be a 3 (fair). This is due to the fact that it is difficult to determine if the deformation is definitely caused by slope instability or is due to other factors.

Step 2: calculation of failure mode indices

Table 6.3 shows the failure mode indices calculated. The table shows that 'slope instability' is the FI with the highest value indicating that this is the

Fig. 6.9 Rutting of embankment crest and slope caused by third party (livestock).

Fig. 6.10 Rutting of slope and crest caused by livestock.

Fig. 6.11 Foreign object presence causing erosion in slope.

most likely mode of failure based on the visual inspection.

Step 3: calculation of condition index

The asset CI calculated for this example using the **weakest FI** ('slope instability' in this case) approach is as follows:
Asset CI = 3.25
Minimum CI = 3.00
Maximum CI = 3.25
The asset CI value is equal to the maximum CI calculated using the confidence assessment indicating that performance could be greater than that assigned. Based on the confidence assessment, it also means that the performance should not be any worse than that assigned from the visual inspection. The minimum CI is 3.00, which is determined as the maximum value of the set of minimum FI values. The 3.00 refers to a different failure mode (overtopping leading to breach) than the failure mode referred to by the asset CI and the maximum CI (slope instability). This indicates that not only is the asset in poor condition but this poor condition refers to a number of failure modes and not just a single mode of failure.

Site-Based Trials

The method described here was adapted and employed in a large-scale trial as part of the Thames Estuary 2100 (TE2100) project. This project was concerned with long-term planning and management for flood risk in the Thames Estuary. Over

Table 6.2 Performance Feature Assessment for the Case Study

Performance features	Condition	Confidence	Condition max.	Condition min.
1 Animal burrowing/vermin infestation	2	0	2	2
2 Foreign objects in the crest or rear slope concentrating the erosion process	3	0	3	3
3 Cracking and/or fissuring	3	0	3	3
4 Third-party damage (cattle, vehicles, etc.)	4	0	4	4
5 Direct evidence of seepage or piping	1	0	1	1
6 Visible deformation of cross-section caused by slope instability	4	−1	4	3
7 Erosion of cross-section	3	0	3	3
8 Vegetation condition (outer slope)	2	0	2	2

Table 6.3 Failure Mode Indices with Associated Confidence Range Values

Failure mode index	Slope instability	Piping	Backfill washout	Overtopping leading to breach
Assigned condition	3.25	1.90	2.25	3.00
Minimum confidence	2.30	1.80	1.80	3.00
Maximum confidence	3.25	1.90	2.25	3.00

337 km of defence assets, representing a wide range of asset types and conditions, were inspected covering the whole of the Thames Estuary region.

Results from the use of the method by a number of experienced and non-experienced inspection staff were very promising. The method was found to be much more consistent in the attribution of asset condition. There was little difference between assessments produced by experienced and non-experienced staff – a significant improvement over the current inspection method.

Overall asset condition was also found to be assessed as being in better condition than with the existing inspection method. This was to some extent expected since the method is focused on assessing likely performance based on visual condition rather than purely the visual condition. It is, however, a crucial and significant vindication of the work. A primary motivation for the project was the hypothesis that the existing inspection method graded asset condition more severely than was necessary due to its focus purely on visual condition without explicit linkage to likely performance. However, this improvement in asset condition will need further research and trials to establish its accuracy.

Critical feedback relating to the complexity of inspecting composite assets and specific concerns on individual flowcharts were also recorded and will be used to further refine the method.

Overall, the results of this initial live trial were encouraging and will provide feedback and experience for any future implementation of the method nationally by the Environment Agency.

Summary

The current method for the visual inspection of flood defence assets in the UK does not realistically assess the likely performance of an asset under loading and is overly concerned with assessing purely visual deterioration, which might not significantly impact on performance.

A revised method has been described that represents a measured step towards a more performance-based assessment using visual findings. This method draws upon recent research in performance modelling and includes uncertainty assessment and potential failure modes for key asset types. The method is backed up through flowchart-based guidance and has been developed in conjunction with leading industry practitioners and researchers.

A number of case studies utilizing well-known assets were developed to provide proof of concept, and the method has been adapted and employed in large-scale live trials as part of the TE2100 project. Results from case studies and the live trial have been very encouraging, and it is envisaged that the method will form the basis for future inspection of assets in the UK.

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Part 3
Flood Forecasting and Warning

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7 Advances in the Remote Sensing of Precipitation Using Weather Radar

IAN D. CLUCKIE

Introduction

Remote sensing has included the application of radar technology in its array of techniques from earliest times. Many of the early developments were stimulated by the needs of war and the military requirement to detect ships, aeroplanes and missiles (see Austin and Cluckie 2006). The hydrological community has also influenced the measurement of precipitation using quantitative weather radar technology, with meteorologists, atmospheric scientists, physicists and electrical engineers contributing their part in what is a multi-disciplinary area of development. Currently, there is interest in exploiting multi-parameter radars that utilize dual-polarization technology to improve the estimation of precipitation. This chapter discusses some of the advantages and disadvantages of the present approaches to precipitation estimation that use both single- and dual-polarization weather radar measurements and provides guidance on future developments.

One of the main advantages of weather radars is that they can scan large spatial precipitation domains from a single location in real time. Networks of raingauges that could measure the same area with a similar spatial resolution are impractical. However, weather radars have practical limitations and the estimation of precipitation using radar is prone to several sources of error. Researchers have described many of the problems that weather radars face in quantitative precipitation estimation (see Atlas and Ulbrich 1990) and in order to obtain an accurate estimate of precipitation, it is necessary to understand each physical process involved.

Weather radar echoes were originally considered as noise to be removed, rather than something of intrinsic interest (Probert-Jones 1990). The original work on radar propagation was done at the Telecommunication Research Establishment [later known as the Royal Signals and Radar Establishment (RSRE) at Malvern, UK] and at the General Electric Research Laboratories by Ryde and colleagues (see Probert-Jones 1990). Ryde and his team estimated the echo intensity and attenuation by atmospheric hydrometeors at centimetric wavelengths to determine the extent to which radars would be affected by bad weather. By 1946 this had established the basis of radar meteorology (Atlas and Ulbrich 1990; Probert-Jones 1990). Marshall *et al.* (1947) were the first to show a strong correlation between the received echo power from a 10-cm wavelength radar and the reflectivity, Z, computed from the drop size distribution of samples obtained from a filter paper on the ground. They stated: 'it may be possible therefore to determine with useful accuracy the intensity of rainfall at a point quite distant (say 100 km) by the radar echo from that point'. This was a seminal statement that underlies all modern weather radars, and this work has stimulated many researchers to estimate precipitation using a radar as a remote sensor of precipitation.

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Multi-Parameter Radar and Dual-Polarization Measurements

Radar generates a pulse of high-power microwave (electromagnetic) energy in some specific direction. This pulse travels at the speed of light and if a reflective target (e.g. precipitation particles) lies along the path of the beam, then a small percentage of the energy is reflected back (also at the speed of light) to the radar and collected by the antenna. This backscattered energy can then be related to the rainfall rate using empirical equations. Its location can be determined from the direction of the radiated pulse and the time it took to travel from the radar to the target and back again. In order to fully exploit and interpret the backscattered electromagnetic radiation from hydrometeors, it is necessary to take into account four fundamental properties of electromagnetic waves: the amplitude, phase, frequency and polarization (Jameson and Johnson 1990) of the signal. The use of the reflectivity factor Z_h exploits the amplitude property and has been the most important parameter in the estimation of precipitation from the earliest days of weather radar development. However, there are several sources of uncertainty using the reflectivity factor that can be minimized by the use of dual-polarization techniques. The dual-polarization measurements are sensitive to size, shape, orientation and phase of the hydrometeors (Herzegh and Jameson 1992) and can in principle improve the estimation of precipitation from weather radars (Zrnic and Ryzhkov 1999). However, the practical realization of the potential from these additional measurements can be limited by the electromechanical problems inherent in current dual-polarization radar technology.

Dual-polarization weather radars alternately transmit vertically and horizontally polarized electromagnetic waves and receive polarized backscattered signals. The backscattering characteristics of a single precipitation particle are described in terms of the backscattering matrix **S** (Bringi and Chandrasekar 2001). The dual-polarization radar measurements (Bringi and Hendry 1990) are related to the scattering elements of the backscattering matrix and they are defined below.

Reflectivity factor

The reflectivity factors at horizontal and vertical polarizations are given by:

$$Z_{h,v} = \frac{\lambda^4}{\pi^5|K|^2}\int \sigma_{h,v}(D)N(D)dD \qquad (7.1)$$

where D is the drop diameter, λ is the radar wavelength, $|K|^2$ is the refractive index of the hydrometeors (approximately 0.93 for water and 0.20 for ice), $\sigma(D)$ is the backscattering cross-sections of the scatterers and $N(D)$ is the drop size distribution. If the scatterers are considered to be water spheres with small radii when compared to the radar wavelength, then the approximation due to Rayleigh applies and Equation 7.1 can be expressed as:

$$Z = \int D^6 N(D)dD \qquad (7.2)$$

Differential reflectivity

Pruppacher and Beard (1970) found that large raindrops falling to the ground are generally distorted into oblate spheroids due to aerodynamic forces. Their maximal dimensions are horizontally oriented even when turbulence, drop collisions and aerodynamic instability may disturb their orientation. Taking advantage of this, Seliga and Bringi (1976) proposed the use of differential reflectivities at orthogonal polarizations to improve the estimation of precipitation. The backscattering cross-section for raindrops is larger for a horizontal polarized wave than for a vertical polarized wave and the differential reflectivity is given by:

$$Z_{dr} = 10\log\left(\frac{Z_h}{Z_v}\right) \qquad (7.3)$$

Seliga and Bringi (1976) showed that the mean volumetric diameter of raindrops is related to the value of Z_{dr}. Therefore Z_{dr} is a measure of the mean particle shape and large raindrops produce large values of Z_{dr}. The sensitivity of Z_{dr} to particle shape is less for ice than for water (Herzegh and Jameson 1992), and ice particles tend to wobble and spin in their descent resulting in Z_{dr} values being closer to zero.

Specific differential phase

Electromagnetic waves experience phase shifts as they propagate through regions of precipitation. The horizontally polarized wave suffers larger phase shifts than the vertically polarized wave because raindrops are horizontally orientated as they fall. The differential phase Φ_{dp} is the difference between the received phases of horizontally and vertically polarized electromagnetic waves ($\Phi_{dp} = \Phi_{hh} - \Phi_{vv}$), and the specific differential phase (K_{dp}) is the rate of change of Φ_{dp} along the range and it is then given by:

$$K_{dp} = \frac{1}{2}\frac{d\Phi_{dp}}{dr}\ {}^{\circ}\mathrm{km}^{-1} \tag{7.4}$$

It can also be expressed as (Bringi and Chandrasekar 2001):

$$\begin{aligned} K_{dp} &= \left(\frac{180^{\circ}}{\lambda}\right)10^{-3}CW(0.062D_m)\ \text{with} \\ D_m &= \frac{\int D^4 N(D)dD}{\int D^3 N(D)dD} \end{aligned} \tag{7.5}$$

where λ is the wavelength in metres, $C \sim 3.75$, W is the water content in grams per cubic metre, and D_m is the mass-weighted mean diameter in millimetres. Equation 7.5 is important because K_{dp} is almost linearly related to the liquid water content (LWC) multiplied by the mean raindrop shape and therefore it provides the possibility of better estimates of the actual rainfall rate.

Estimation of Precipitation Using Weather Radar

Raindrops grow to a critical size and then suffer break-up due to hydrodynamic instability (Cotton and Anthes 1989). The different sizes of raindrops define a drop size distribution (DSD) given by $N(D)$. The DSD describes the probability density function of the raindrop sizes and it is one of the most important functions in rainfall rate estimation algorithms. All the microphysical processes involved in the interaction between the raindrops are reflected in the DSD, and for a given DSD the rain rate can be obtained from:

$$R = 0.0006\pi \int v(D)D^3 N(D)dD\ \mathrm{mm\,h^{-1}} \tag{7.6}$$

where D is the raindrop diameter and $v(D)$ is the terminal velocity of the raindrops. Atlas and Ulbrich (1977) expressed the terminal velocity as a function of the particle diameter, given by $v(D) = 3.78D^{0.67}\ \mathrm{m\,s^{-1}}$, assuming the absence of vertical air motions. Thus, R represents the 3.67th moment of the DSD whereas Z represents the 6th moment (see Equation 7.2), with Z being more sensitive to large drops than R. Knowledge of the DSD is important because it establishes the interaction between the radar reflectivity and the rainfall rate. Marshall and Palmer (1948) observed an exponential DSD, and the exponential form of the DSD can always be applied when a large number of DSD are averaged in space or time (Bringi and Chandrasekar 2001). The Marshall and Palmer DSD can be expressed as:

$$N(D) = N_0 \exp(-3.67D/D_0) \tag{7.7}$$

where N_0 is 8000 $\mathrm{m^{-3}\,mm^{-1}}$ and D_0 is the median drop diameter, which is defined as the drop diameter such that 50% of the water content comprises drops with diameters less than D_0 (Doviak and Zrnic 1993). According to Marshall and Palmer (1948) the exponential DSD slightly overestimates for raindrop diameters less than 1.5 mm in diameter. However, in reality there are larger variations in the shape of the DSD not represented by the Marshall and Palmer DSD, and Ulbrich (1983) proposed a more general three-parameter gamma DSD given by:

$$N(D) = N_0 D^{\mu} \exp\left(\frac{-(3.67+\mu)D}{D_0}\right) \tag{7.8}$$

where the parameter μ takes values between -3 and 8. For $\mu = 0$, Equation 7.8 takes the form of the Marshall and Palmer DSD. The shape of the gamma DSD is determined by the exponent μ, and for positive values of μ the gamma DSD is concave down whereas for negative values it is concave upward. A gamma DSD can describe many of the natural variations in the shape of the DSD. When there is a substantial depth between the melting level and the ground surface, the parameterization of a gamma DSD appears to be suitable for stratiform and convective rainfall events (Bringi and Chandrasekar 2001). In addition, Testud *et al.* (2001) proposed the normalization of the DSD to avoid any assumption about the shape of the DSD.

Algorithms to Estimate Rain from Radar Measurements

The most commonly used polarimetric radar measurements for rainfall estimation are the reflectivity factor (Z_h), the differential reflectivity (Z_{dr}) and the specific differential phase (K_{dp}), and for many years radar meteorologists have tried to find a useful equation relating the reflectivity factor to the rainfall rate. The rainfall rate given by Equation 7.6 can be obtained by assuming a drop size distribution and terminal velocity of the raindrops. By comparing the rainfall rate with the actual reflectivity measured by the radar, it is possible to derive *Z-R* relationships of the form:

$$Z = aR^b \text{ or } R = a^{-1/b} Z^{1/b} \tag{7.9}$$

where Z is the reflectivity factor in $\mathrm{mm^6\,m^{-3}}$, R is the rainfall rate in $\mathrm{mm\,h^{-1}}$, and a and b are the parameters obtained from a regression analysis. Atlas and Ulbrich (1990) showed that the first *Z-R* relationship could be traced back to the research work carried out by Ryde (1946). They also showed that this relationship is approximately $Z = 320R^{1.44}$. This is very similar to that employed to estimate the rainfall rate from reflectivity measurements in the WSR-88D radar network, which is $Z = 300R^{1.4}$ (Serafin and Wilson 2000). Marshall *et al.* (1947) reported one of the first *Z-R* relationships ($Z = 190R^{1.72}$), which was later slightly modified to $Z = 220R^{1.6}$ (Marshall and Palmer 1948). Some years later, Marshall *et al.* (1955) slightly revised the 1948 relationship, obtaining the well-known Marshall–Palmer formula $Z = 200R^{1.6}$. The UK Meteorological Office's Nimrod system estimates the precipitation rate using this equation. Unfortunately, there is no single *Z-R* relationship that can be applied in every part of the world. Battan (1973) listed 69 different *Z-R* relationships derived from different climatological regions by many researchers. This variability was due to the fact that the coefficient and exponent of the *Z-R* relationship depend on the shape of the DSD. Therefore, it is necessary to estimate in real time the parameters of the DSD to allow flexibility in the variation of the parameters a and b of the *Z-R* relationship.

One of the main goals of dual-polarization radars is the improvement in quantitative precipitation estimates, and Seliga and Bringi (1976) proposed the use of differential reflectivities at orthogonal polarizations to estimate the parameters of an exponential DSD (Equation 7.7). They suggested that the parameter D_0 is obtained with Z_{dr} whereas N_0 is obtained with Z_h and D_0. The main advantage of using the differential reflectivity Z_{dr} is that the median raindrop diameter D_0 is related to the value of Z_{dr}. Using the differential reflectivity measurements has been exploited by several researchers to obtain a more representative shape of the DSD.

In order to derive relationships between the rainfall rate and the polarimetric radar measurements a common method is based on varying the parameters N_0, D_0 and μ of a theoretical gamma DSD and then calculating R, Z_h, Z_{dr} and K_{dp} assuming a scattering model. In addition, the mean raindrop axis ratio $r(D)$, or degree of deformation as a function of the diameter D, must be specified. The coefficients and exponents of the different rainfall rate algorithms are usually obtained by performing a non-linear regression between the rainfall rate and the polarimetric variables (Bringi and Chandrasekar 2001). The deformation of raindrops is an important relationship that leads to different rainfall

Table 7.1 Summary of relationships for rainfall estimation using polarimetric radar measurements at different frequencies f. R is in mm h^{-1}, Z_h is in mm^6 m^{-3}, Z_{dr} is in dB, K_{dp} is in degrees per kilometre and λ in cm

Rain estimator	f (GHz)	c	a	b	
$R = a^{-1/b}Z^{1/b}$	> 3	–	200	1.6	Marshall *et al.* (1955)
$R = cZ_h^a 10^{0.1bZdr}$	3	0.0067	0.93	−3.43	Bringi and Chandrasekar (2001)
	5	0.0058	0.91	−2.09	Bringi and Chandrasekar (2001)
	10	0.0039	1.07	−5.97	Bringi and Chandrasekar (2001)
$R = cK_{dp}^b$	3–10	$5.1\lambda^{0.86}$	–	0.866	Sachidananda and Zrnic (1987)
	3	50.7	–	0.85	Bringi and Chandrasekar (2001)
	3	40.5	–	0.85	Bringi and Chandrasekar (2001)
	3	50.1	–	0.70	Illingworth (2003)
	5	31.4	–	0.70	Illingworth (2003)
$R = cK_{dp}^a 10^{0.1bZdr}$	3	90.8	0.93	−1.69	Bringi and Chandrasekar (2001)
	5	37.9	0.89	−0.72	Bringi and Chandrasekar (2001)
	10	28.6	0.95	−1.37	Bringi and Chandrasekar (2001)

estimators, but further research has to be done to reach a consensus on raindrop deformation.

Gorgucca *et al.* (1994) proposed an algorithm to estimate the rainfall rate based on Z_h and Z_{dr} (Table 7.1). The disadvantage of the algorithm $R(Z_h, Z_{dr})$ is that Z_h and Z_{dr} are prone to attenuation of the horizontal reflectivity and differential attenuation, respectively, at frequencies higher than 3 GHz, quite apart from the fact that Z_h can be subject to radar miscalibration. The use of the differential phase K_{dp} may overcome these difficulties but the main advantages of rainfall estimation based on K_{dp} are its immunity to attenuation by precipitation, its immunity to radar miscalibration, and the fact that K_{dp} is also less affected by partial blockage of the radar beam (Zrnic and Ryzhkov 1996). Thus, rainfall estimators based on K_{dp} (Sachidananda and Zrnic 1987) and K_{dp}-Z_{dr} (Bringi and Chandrasekar 2001) have been proposed (Table 7.1).

It is clear that the different algorithms to estimate the rainfall rate have advantages and disadvantages. Relationships of the form $R(Z_h)$ have been used from just after World War II. However, this type of relationship contains uncertainty in the coefficients a and b because they are related to the shape of the DSD. Without extra information, apart from the reflectivity factor, a and b have to be obtained empirically by establishing a single climatological Z-R relationship. Even if the DSD is known, the $R(Z_h)$ relationship is still critically dependent on the calibration of the radar system. To avoid any bias in the measurement of Z_h, it is necessary to calculate accurately the radar constant. In addition, Z_h is not immune to propagation effects and it is subject to attenuation due to rain at frequencies higher than 3 GHz.

Relationships involving Z_{dr} are also questionable because although Z_{dr} is independent of radar calibration, it is not immune to propagation effects, being subject to differential attenuation in heavy precipitation and the depolarization of the polarized waves. Illingworth (2003) discussed the accuracy of rainfall estimates using Z_h and Z_{dr}. He argued that the accuracy of $R(Z_h, Z_{dr})$ depends on several factors such as the accuracy of Z_{dr} to 0.2 dB for $R > 10$ mm h^{-1} and less for lower rainfall rates. In practice, this is very difficult to achieve because there are other factors limiting the accuracy of Z_{dr}. For instance, Z_{dr} may be contaminated by the power of the sidelobes of the beam radiation pattern due to reflectivity gradients, and it may also be affected by the mismatch between the horizontal and vertical beam radiation patterns causing the sampling of different volumes of precipitation (Illingworth 2003).

However, relationships of the form $R(K_{dp})$ present several advantages as mentioned previously but Φ_{dp} is extremely noisy and consequently K_{dp} will be even noisier (see Equation 7.4). To decrease the noise, Φ_{dp} is averaged for several kilometres along the beam. Ryzhkov and Zrnic (1996) have even suggested averaging in range over 2.5 km and 7.5 km approximately (17 and 49 gates, respectively, with a resolution of 150 m) for a threshold in reflectivity of 40 dBZ. This obviously leads to a considerable loss in resolution over the conventional $R(Z_h)$ rainfall estimator. Brandes *et al.* (2001) carried out an analysis between raingauge observations and rainfall rates estimated from K_{dp} and Z_h, and they found similar bias factors and correlation coefficients between both estimators, concluding that no obvious benefit is obtained in using K_{dp} to estimate rainfall rates over using Z_h from a well-calibrated radar.

Therefore, there is controversy whether or not polarimetry is going to improve radar rainfall estimates. Illingworth (2003) suggested that at the 2-km scale needed for an operational environment, the additional information provided by Z_{dr} and K_{dp} is not sufficiently accurate to improve rainfall estimates. However, some improvement in the precipitation estimates from polarimetric radar measurements may be realized by not only applying one particular rain estimator, but also exploiting the attributes of the many different polarimetric algorithms available depending on the circumstance. Ryzhkov and Giangrande (2004) proposed a 'synthetic' algorithm, which makes use of different combinations of rain rate algorithms depending on the rain rate estimated using only the conventional $R(Z)$ relationship. They proposed the use of an algorithm of the form $R(R(Z_h), Z_{dr})$ for low rain rates ($R < 6\,\mathrm{mm\,h^{-1}}$), of the form $R(R(K_{dp}), Z_{dr})$ for medium rain rates ($6 < R < 50\,\mathrm{mm\,h^{-1}}$), and the algorithm $R(K_{dp})$ for high rain rates ($R > 50\,\mathrm{mm\,h^{-1}}$). Although the algorithms proposed by Ryzhkov and Giangrande (2004) are slightly different from the rainfall rate estimators presented in this section, it is clear that by exploiting the performance of different relationships $R(Z_h)$, $R(Z_h, Z_{dr})$, $R(K_{dp})$ and $R(K_{dp}, Z_{dr})$, it may be possible to improve the estimation of rainfall using dual-polarization radars. However, the move from research radars into the operational domain is still to be proven on the basis of sound operational experience.

In addition, polarimetric radar measurements offer the possibility to classify hydrometeors (Zrnic and Ryzhkov 1999; Vivekanandan *et al.* 1999; Liu & Chandrasekar 2000; Zrnic *et al.* 2001), which provides the possibility of applying different rainfall estimators depending on the classification. However, the operational performance of such radars in practice is again still to be established.

Problems Associated with the Estimation of Precipitation

The estimation of precipitation using weather radars is subject to a variety of error sources. In the previous section the importance of the DSD has been described in relating the reflectivity factor Z_h (or any of the polarimetric variables Z_{dr}, K_{dp}) to the rainfall rate R. However, uncertainties in the knowledge of the DSD may not be the largest source of errors in radar rainfall measurements (Joss and Waldvogel 1990) and there are additional errors that may require even more attention. Atlas *et al.* (1984) concluded that the average deviation in the rain rate estimation from reflectivity measurements due to DSD variability would be 33%, whereas Doviak and Zrnic (1993) suggest errors of 30–35%. However, Joss and Waldvogel (1990) suggest that after averaging over space and time, the errors in rainfall estimates due to the variability of the DSD rarely exceed a factor of two. Problems associated with the variation of the vertical reflectivity profile of precipitation may be one of the largest sources of error (Fig 7.1). As the range increases from the radar, the radar beam is at some height above the ground. The hydrometeors intercepted by the radar beam may then be composed of raindrops, melting snowflakes, snowflakes, hail, etc. This variability affects reflectivity measurements and the estimation of precipitation is not representative of the rainfall rate at the ground. This variation is due to the growth or evaporation of precipitation, or change of phase, in particular melting, where a layer of

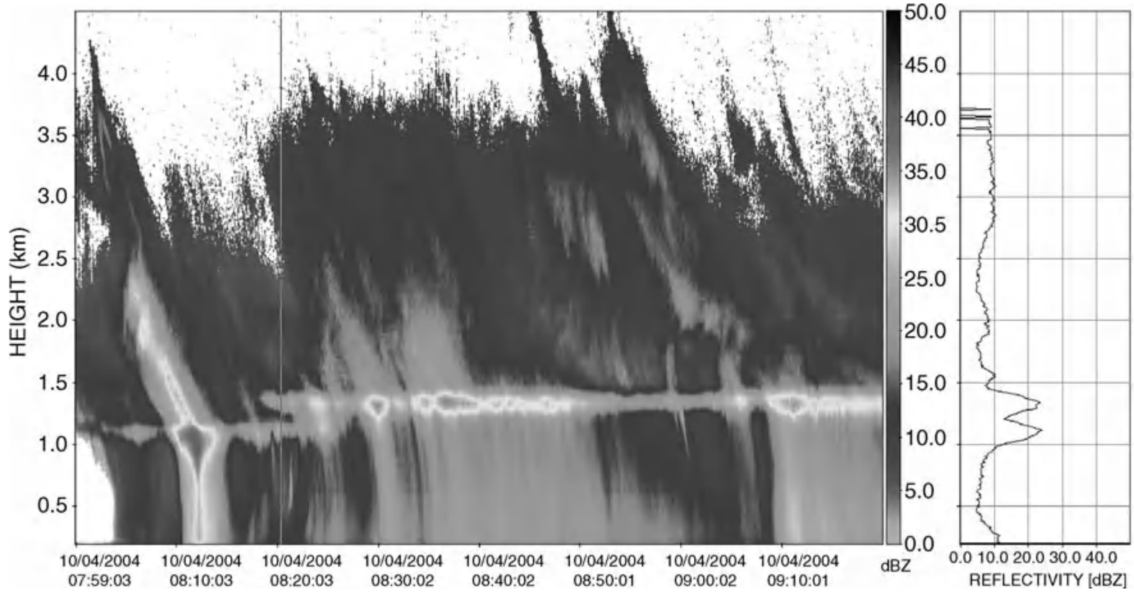

Fig. 7.1 Variation of the vertical reflectivity profile of precipitation. The data were obtained with a Vertically Pointing Radar (VPR) (see Cluckie *et al.* 2000). (See the colour version of this figure in Colour Plate section.)

enhanced reflectivity caused by melting snowflakes produces errors by up to a factor of five. Various correction algorithms have been proposed to correct for the variation of the Vertical Reflectivity Profile (VRP) (Tilford 1992; Hardaker 1993; Fabry 1994; Kitchen *et al.* 1994; Gray *et al.* 2002; Rico-Ramirez 2004), but further research is required to fully account for the variation of the melting level in operational applications.

A feature is observed when temperature inversions exist along the vertical. As ice particles fall through an upper 0°C isotherm layer, they start melting, then refreezing at another level of subzero temperature until melting starts again. The result, when observed by a VPR, is a double bright band (see reflectivity plot to the left of Fig. 7.1). Double bright bands are relatively common in fronts, which tend to contain temperature inversions. The problem for scanning radars is that the effect is dynamic **and** can occur anywhere within the radar image.

Partial blockage of the beam is especially problematic in hilly terrain. The radar generally scans at low elevation angles to obtain measurements close to the ground. Ground echoes from nearby mountains can be misinterpreted as heavy precipitation and therefore overestimation may occur. Correction for partial beam blockage is difficult because the power is reflected back to the radar not only from the main lobe but also from the secondary sidelobes. By knowing the characteristic of the antenna radiation pattern it is sometimes possible to apply a correction for partial blocking of the radar beam, but in conditions where the beam departs slightly from the standard propagation pattern it is not so straightforward. The use of K_{dp} for rainfall estimation may help in overcoming problems due to partial beam blocking, but as mentioned in the previous section, K_{dp} is very noisy and it is only useful in very heavy precipitation. Therefore, it is important to establish accurate corrections to overcome the effects of partial blockage of the beam.

Attenuation by precipitation is another source of error, especially at frequencies higher than 3 GHz. To some extent this is now recognized as posing a problem at C-band frequencies as well as X-band. It has been shown that the attenuation is directly proportional to the rain rate R, and expressions of the form $A = aR^b$ have been obtained. Attenuation correction algorithms have been developed using the specific differential phase (Bringi *et al*. 2001) when the radar beam passes through rain-filled media, but additional research has to be done to correct for attenuation when the radar beam passes through melting snow or mixed-phase precipitation.

Conclusions

Many research challenges remain to be overcome to increase the reliability of single- and dual-polarization radar measurements to predict rainfall. The greatest benefits are likely to come from work to account for the variation of the vertical reflectivity profile, in particular when the melting layer is at lower altitudes. This may not be a problem in regions where the melting level is at higher altitudes, but it is a real problem in regions such as the UK. Polarimetric radar measurements offer the possibility of classifying hydrometeors, which allow the application of different rainfall estimators and attenuation corrections within the rain region. Difficulty still remains in estimating rainfall rates in snow and melting snow, and polarimetric radar measurements potentially provide advantages over conventional reflectivity radars in discriminating hydrometeor types. However, these advantages are still to be realized operationally in the context of precipitation estimation from weather radar and their quantitative use in hydrology. Hydrologically, the concentration of future effort on 'flood-producing' storms over urban areas will also focus research effort on the issue of measurements at both space and time scales appropriate for applications over large urban areas.

Acknowledgement

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8 Artificial Intelligence Techniques for Real-Time Flood Forecasting

JONATHAN LAWRY, DANIEL R. McCULLOCH, NICHOLAS J. RANDON AND IAN D. CLUCKIE

Abstract

Fuzzy rule-based artificial intelligence techniques for flood forecasting are introduced. These incorporate both fuzziness and probabilistic uncertainty, which are inherent in many hydrological systems. Two algorithms, Linguistic Decision Trees and Fuzzy Bayes, are applied in a number of case studies relating to weather radar image classification and flow and level time series forecasting. The models produced are shown to be transparent and understandable as well as providing accurate forecasts.

Introduction

Artificial intelligence (AI) algorithms such as those developed in the field of machine learning are a source of powerful new techniques for flood prediction. By generating models automatically from data they can provide computationally efficient and accurate prediction models for real-time forecasting that avoid the costs of solving complex non-linear equations. As such they can form the basis of new prediction tools, which can be run locally on relatively low-specification computers, complementing more traditional methods.

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Traditional approaches to flood forecasting involve multi-dimensional mathematical models based extensively on underlying physical principles. In contrast, machine learning algorithms are data-driven methods whereby models are inferred directly from a database of training examples. Consequently the incorporation of background knowledge, in the form of an understanding of the hydrology of the system being studied, only takes place indirectly through, for example, the choice of input variables to the AI algorithm, or through the identification of an appropriate lead time for prediction. For this reason data-driven models are sometimes referred to as being 'black box'.

Typical examples of machine learning algorithms that have been applied to flood prediction are Neural Networks (NN) and Support Vector Machines (SVM). Both methods are based on the generation of separating hyperplanes, for the former in attribute space and for the latter in non-linear transformation of attribute space. Examples of Neural Network applications include that of Campolo *et al.* (1999), which applied NNs to the prediction of water levels for the River Tagliamento in Italy, and that of Han *et al.* (2007a), which compared NN to transfer function models. Recent research into forecasting based on SVMs includes Han *et al.* (2007b) and Randon *et al.* (2004) on the Bird Creek catchment in USA, and Yu *et al.* (2006) on the Yan-Yang river basin in Taiwan.

Algorithms such as NNs and SVMs are also black-box in a different way, in that the models they generate are very difficult to interpret.

Although they may give accurate predictions, they provide little or no insight into the underlying nature of the system being modelled. Furthermore, it is difficult to identify a transparent set of conditions that result in a particular prediction. This makes it difficult to trace and examine the basis for decisions informed by tools that incorporate black-box methods. In this chapter we shall instead focus on the application of rule-based methods to flood forecasting. These rules incorporate linguistic descriptions of values in the form of fuzzy labels and also allow for the explicit representation of probabilistic uncertainty. Hence, for a particular prediction they enable us to identify a set of fuzzy mappings from which the prediction is generated. In summary, we will make the case that AI methods based on fuzzy-probabilistic rules can potentially provide an effective and transparent tool for flood forecasting.

In the next section we introduce Label Semantics, an integrated theory of fuzziness and probability. Following this we describe two fuzzy learning algorithms, Linguistic Decision Trees and Fuzzy Bayes. The next two sections present case studies of applications in flood forecasting: the first shows how linguistic decision trees can be used to classify regions of weather radar images so as to aid in Bright Band detection; the second introduces a number of problems involving time-series modelling including flow forecasting for the Bird Creek catchment (USA) and level prediction for the Upper Severn catchment (UK). Finally, we provide some discussion and conclusions.

Integrating Fuzzy and Probabilistic Uncertainty

Both uncertainty and fuzziness are inherent to the complex systems associated with flood forecasting. The former results from natural random processes, from model incompleteness and from a lack of information about important parameters and measurements. The latter is a consequence of imprecision and noise in data measurements including river levels, river flow, rainfall etc. **Label semantics** has been introduced by Lawry (2006) as an uncertainty theory for fuzzy description labels, which allows for an integrated treatment of both types of uncertainty within a coherent unified framework.

In label semantics a continuous universe Ω is partitioned using a finite set of labels LA (e.g. possible labels include *low, medium, high, about 20*, etc.). Each label L is defined by an appropriateness measure $\mu_L : \Omega \rightarrow [0, 1]$ where for $x \in \Omega$ $\mu_L(x)$ is the subjective probability that label L is an appropriate label to describe x. Figure 8.1 shows trapezoidal appropriateness measures for labels $L_1, \ldots, L_n$. In this case each label describes a closed interval of the real line together with a neighbouring region, with decreasing appropriateness as distance from the core interval increases.

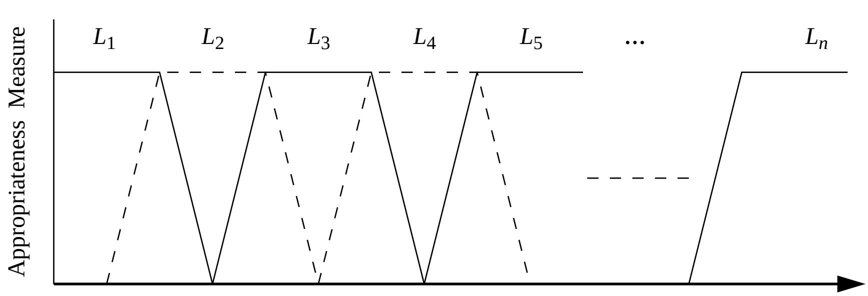


Fig. 8.1 Fuzzy labels defined by trapezoidal appropriateness measures.

In addition to appropriateness measures a second related measure can be defined for the fuzzy labels in LA. For value $x \in \Omega$ the mass function $m_x : P(LA) \rightarrow [0, 1]$ represents the uncertainty about which set of labels is appropriate to describe x. For example, $m_x(\{low, medium\})$ is the probability that *low* and *medium* are both appropriate to describe x and all other labels are inappropriate. This mass value would tend to be high for values of x where both *low* and *medium* are highly appropriate and hence where the two appropriateness measures overlap. The mass function m_x is a probability distribution on the power set of labels $P(LA)$ so that:

$$\sum\nolimits_{F \subseteq LA} m_x(F) = 1 \tag{8.1}$$

The strong relationship between mass functions and appropriateness measures is based on the fact that the probability that label L is appropriate to describe x is equivalent to the probability that label L is included in the set of labels appropriate to describe x; consequently we have the relationship:

$$\mu_L(x) = \sum\nolimits_{F:L \in F} m_x(F) \tag{8.2}$$

Furthermore, Lawry (2006) shows that, assuming labels can be ranked in terms of their appropriateness, mass functions can be determined directly from appropriateness measures. Figure 8.2 shows the mass functions derived from the trapezoidal appropriateness measures in Figure 8.1.

Learning Algorithms

This section gives a brief description of two learning algorithms, Linguistic Decision Trees and Fuzzy Bayes, based on the Label Semantics framework introduced in the previous section. These will be the core AI technologies applied in the hydrological case studies given in the following two sections.

Linguistic decision tree

Linguistic Decision Trees (LDTs) have been proposed by Lawry (2006) as a tree-structured representation for conditional rules involving fuzzy labels. LDTs consist of nodes corresponding to input variables (attributes) and branches corresponding to linguistic expressions describing variables in terms of a set of predefined fuzzy labels. Associated with each complete branch there is a probability distribution. In the case of a classification model this will be the probabilities for the different classes. For a prediction model where the output is a real value then the probabilities define a mass function on the label sets describing that value.

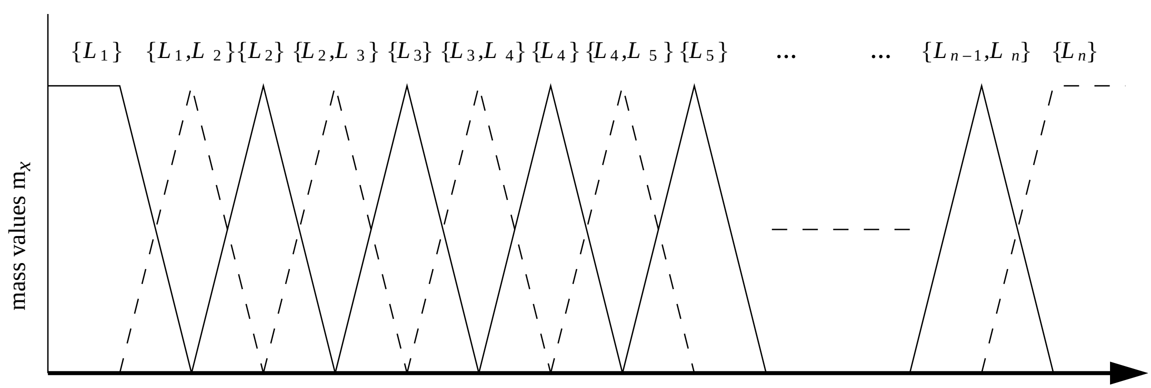


Fig. 8.2 Mass function for the possible sets of appropriate labels derived from the appropriateness measures in Figure 8.1.

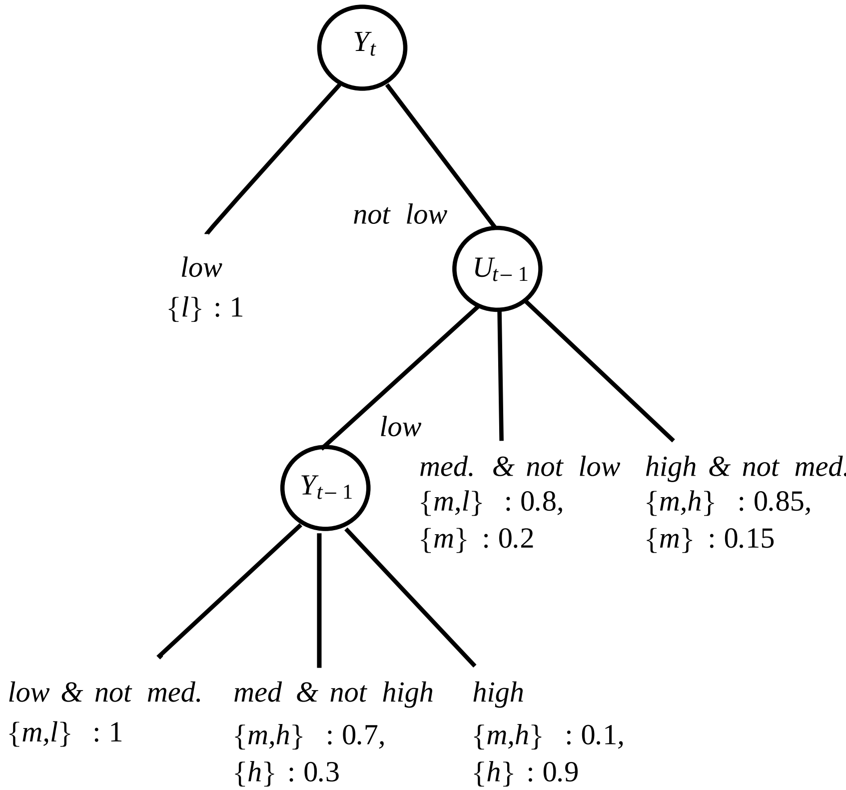


Fig. 8.3 Hypothetical linguistic decision tree (LDT) mapping current and historical flow and rainfall measurements to predicted future flow measurements. See text for definitions of labels.

Consider a time series consisting of flow Y_t and rainfall measurements U_t. Figure 8.3 shows a hypothetical LDT relating current and historical flow and rainfall measurements to predicted future flow $Y_{t+\delta}$. Here it is assumed that both flow and rainfall measurements are described by labels *low (l)*, *medium (m)* and *high (h)* although these may be defined according to differently scaled appropriateness measures. The mass function associated with each branch refers to the set of labels appropriate to describe $Y_{t+\delta}$ given the conditions on the other variables identified in that branch. Hence each branch represents a probabilistic rule based on fuzzy labels. For example, the branch labelled B in Figure 8.3 represents the following rule:

- IF Y_t is *not low* AND U_{t-1} is *low* AND Y_{t-1} is *medium but not high* THEN $Y_{t+\delta}$ is either *between medium and high* (with probability 0.7) or *only high* (with probability 0.3).

Qin (2005) and Qin and Lawry (2005) introduced the Linguistic ID3 LID3 algorithm to learn LDTs from data. LID3 is an extension of the well-known ID3 algorithm (Quinlan 1986) to generate probabilistic decision trees involving fuzzy labels. Shannon's entropy measure is used as a search heuristic in the generation of the LDT so that branches are expanded using attributes that minimize the uncertainty (entropy) associated with the resulting branch probability distributions. LID3 has been applied to a wide range of problem areas including radar image classification and river level forecasting as described below. See Qin (2005) and Qin and Lawry (2005) for discussions on the application of LID3 to many different classification and prediction problems.

The Fuzzy Bayesian approach

The Fuzzy Bayesian approach was proposed in Randon and Lawry (2002) and Randon (2004) as a Label Semantics based method for combining Bayesian learning algorithms with fuzzy labels. For classification problems the standard Bayesian approach determines the conditional probability of input variables $x_1,\ldots, x_n$ given each class C_i. Bayes' theorem is then applied to determine the probability of each class given instantiations of $x_1,\ldots, x_n$:

$$P(C_i|x_1,\ldots,x_n) = \frac{p(x_1,\ldots,x_n|C_i)P(C_i)}{\sum_i p(x_1,\ldots,x_n|C_i)P(C_i)} \quad (8.3)$$

Here $P(C_i)$ is the proportion of examples in the training data that are of class C_i.

The Fuzzy Bayesian approach applies Bayes' theorem to mass functions instead of standard probability distributions. A set of labels LA_i is defined for each variable x_i and (for continuous valued prediction problems) labels LA_y are defined for the output variable y. A multi-dimensional mass function (referred to as a mass relation) is then evaluated from the training data conditional on each set of description labels for y; this is denoted $m(F_1,\ldots,F_n|F_y)$ where $F_i \subseteq LA_i$ for $i=1,\ldots, n$ and $F_y \subseteq LA_y$. Bayes' theorem is then applied to obtain a mass function on the labels for y given label sets describing the input variables:

$$m(F_y|F_1,\ldots,F_n) = \frac{m(F_1,\ldots,F_n|F_y)m(F_y)}{\sum_{F_y} m(F_1,\ldots,F_n|F_y)m(F_y)} \quad (8.4)$$

Here $m(F_y)$ is the mass function for y derived from the training data by aggregating $m_y(F_y)$ across all values of y in the database.

Randon (2004) investigated the use of a Naive Bayes algorithm in this context where all the input variables are assumed to be conditionally independent given an output description so that:

$$m(F_1,\ldots,F_n|F_y) = \prod_{i=1}^{n} m(F_i|F_y) \quad (8.5)$$

In addition, Randon and Lawry (2002) proposed a Semi-Naive Bayes model where the above independence assumption was weakened to allow for groups of dependent variables with independence assumed between each group.

Fuzzy Naive and Semi-Naive Bayes have been applied to a wide range of application areas including vision, medical classification and time series prediction (see Randon 2004). In the flood management domain Fuzzy Naive Bayes has been applied to the prediction of sea levels in the North Sea (Randon *et al.* 2008), and flow prediction for the Bird Creek catchment, as described below. Also, in hydrological applications of weather radar, a Fuzzy Naive Bayes classifier has been applied to classify precipitation and non-precipitation echoes (Rico-Ramirez and Cluckie 2008).

Classification of Weather Radar Images

The quantitative use of radar-based precipitation estimations in hydrological modelling for flood forecasting has been limited due to different sources of uncertainty in the rainfall estimation process. The factors that affect radar rainfall estimations are well known and have been discussed by several authors (Battan 1973; Austin 1987; Doviak and Zrnic 1993; Collier 1996). These include factors such as radar calibration, signal attenuation, clutter and anomalous propagation, variation of the Vertical Reflectivity of Precipitation (VPR), range effects, Z-R relationships, variation of the drop size distribution, vertical air motions, beam overshooting the shallow precipitation and sampling issues among others.

The VPR is an important source of uncertainty in the estimation of precipitation using radars. The variation is largely due to factors such as the growth or evaporation of precipitation, the thermodynamic phase of the hydrometeors, or melting and wind effects. As the range increases from the radar, the radar beam is at some height above the ground, while the radar sampling volume increases and is unlikely to be homogeneously filled by hydrometeors. As an example, the lower part of the volume could be in rain, whereas the upper part of the same volume could be filled with snow, or even be without an echo. This variability affects reflectivity measurements, and the estimation of precipitation may not represent the rainfall rate at the ground. Snowflakes are generally low-density aggregates and when they start to melt they appear as large raindrops to the radar resulting in larger values of reflectivities compared to the expected reflectivity below the melting layer (Battan 1973). This phenomenon is called 'Bright Band', and the interception of the radar beam with melting snowflakes can cause significant overestimates of precipitation up to a factor of five. When the radar beam is above the Bright Band this can cause underestimates of precipitation up to a factor of four per kilometre above the Bright Band (Joss and Waldvogel 1990).

The Bright Band can be seen as the very dark region in Range Height Indicator RHI scans (seen in Fig. 8.4). The power reflected back to the radar is related to the rainfall intensity and therefore the radar beams striking this melting layer of snow causes overestimation of precipitation. Therefore the Bright Band needs to be detected and corrected for. In addition to this, when estimating precipitation intensity, determining which hydrometeors (i.e. Rain and Snow) the beam intersects is crucial to the calculation.

There are several algorithms reported in the literature that use fuzzy logic to classify hydrometeors (e.g. Vivekanandan *et al.*, 1999; Liu and Chandrasekar 2000; Straka *et al.*2000). Also, Rico-Ramirez *et al.* (2005) proposed a robust algorithm to define the membership functions of a hydrometeor fuzzy logic classifier using high-resolution vertical reflectivity profiles from the Chilbolton radar. McCulloch *et al.* (2007) applied the LID3 linguistic decision tree learning algorithm to RHI scans from the Chilbolton radar

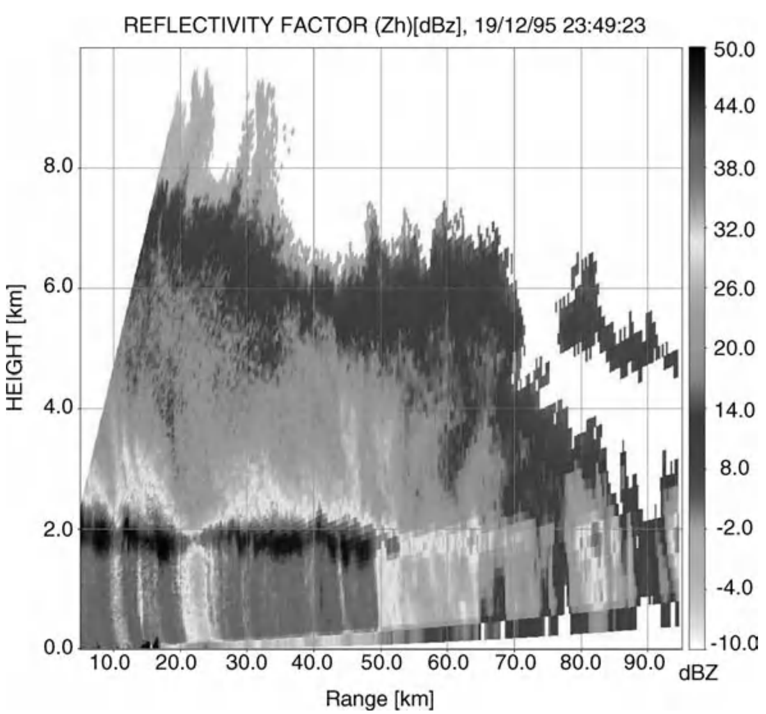


Fig. 8.4 A typical RHI scan for a stratiform event. (See the colour version of this figure in Colour Plate section.)

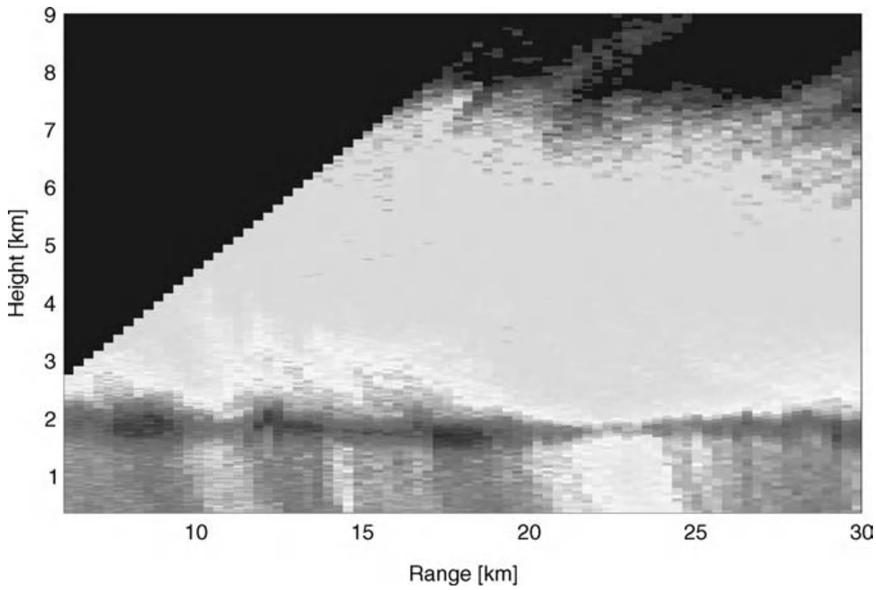


Fig. 8.5 RHI scan from the Chilbolton radar dataset. (See the colour version of this figure in Colour Plate section.)

(Fig. 8.5), an S-band (9.75-cm wavelength) weather radar developed to study the effects of rain on communication systems (Goddard *et al.*1994). The objective was to obtain a set of rules to classify pixels of vertical reflectivity profile images as being either snow, rain or bright band (Fig. 8.6)

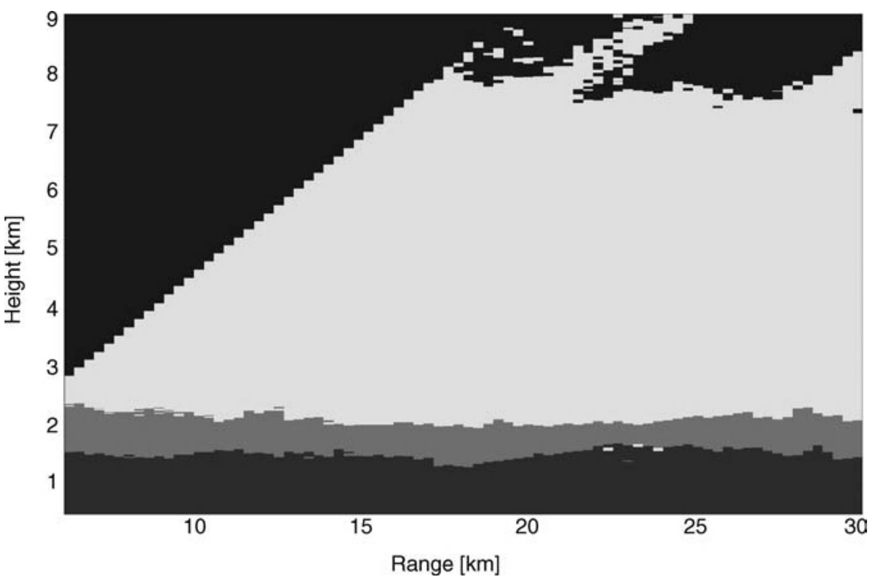


Fig. 8.6 Classification of scan in Figure 8.5 using the LID3 algorithm. Light blue indicates rain, green indicates snow, and red indicates bright band. (See the colour version of this figure in Colour Plate section.)

where the ground truth for the training set corresponded to the class labels allocated by the rotational algorithm proposed by Rico-Ramirez (2004) and Rico-Ramirez and Cluckie (2007). In addition to providing an easily understandable set of robust classification rules, LID3 also has the advantage that it can classify an image in real time, pixel by pixel, in contrast to Rico-Ramirez's algorithm, which requires preprocessing of each image in its entirety before any pixel level classifications can be made. The features used by LID3 were as follows: reflectivity factor (Z_h), the differential refectivity (Z_{dr}), the linear depolarization ratio (L_{dr}) and the height measurement (H_0).

The data for the experiments were generated from 1354 images resulting in 191,235 labelled data vectors. Examples of rules drawn from the resulting linguistic decision tree are as follows:

- IF L_{dr} is between *low* and *med.*, AND H is between *med.* and *high*, AND Z_h is only *high*, AND Z_{dr} is between *med.* and *high* THEN the pixel class is *Rain*: 0.998, *Snow*: 0.002, *Bright band*: 0
- IF L_{dr} is only *med.*, AND H is between *med* and *high*, AND Z_h is between *med.* and *high*, AND Z_{dr} is only *low* THEN the pixel class is *Rain*: 0.03, *Snow*: 0.97, *Bright band*: 0
- IF L_{dr} is only *high*, AND H is only *med.*, AND Z_h is only *high*, AND Z_{dr} is only *high* THEN the pixel class is *Rain*: 0.02, *Snow*: 0, *Bright band*: 0.98

Table 8.1 shows a comparison of the results of LID3 with a number of other machine learning algorithms including Naive Bayes, Neural Networks, Support Vector Machines (SVM) and k-Nearest Neighbour (KNN). The results refer to average percentage accuracy in a 10-fold cross-validation experiment where the algorithms are repeated trained on a sample of $\frac{9}{10}$th of the data and then tested on the remaining $\frac{1}{10}$th.

Table 8.1 Comparison of results with LID3 and Machine Learning Algorithms

	Accuracy			
Algorithm	Rain	Snow	Bright Band	Average
Naive Bayes	75.3%	68.5%	98.6%	80.8%
Neural Network	85.2%	75.8%	99.0%	86.7%
SVM	85.1%	84.0%	98.4%	89.2%
KNN	85.1%	84.0%	98.4%	89.2%
LID3	93.5%	89.0%	99.4%	94.0%

KNN, k-Nearest Neighbour; SVM, Support Vector Machines;

Time Series Modelling

In this section we discuss the application of AI techniques to time series forecasting problems relating to two different river catchments.

Bird Creek catchment

The Bird Creek river basin is in Oklahoma (USA) close to the northern state border with Kansas. The catchment area covers 2344 km^2 with the outlet of the basin near Sperry about 10 km north of Tulsa. The area itself is 175 to 390 m above the mean sea level and has no mountainous regions or large water surfaces to influence local climate condition. Some 20% of the catchment surface is covered by forest and the main vegetative cover is grassland, with the soil storage capacity being described as very high (see Georgakakos and Smith 1990). Figure 8.7 shows the river basin describing the Bird Creek area – see Georgakakos *et al.* (1988) for more information.

The database considered here was collected to form part of a real-time hydrological model intercomparison exercise conducted in Vancouver, Canada, in 1987 and reported by the World Meteorological Organization (1992). The database contains information on average rainfall derived from 12 raingauges situated in or near the catchment area (U_t average rainfall at time t) and on streamflow measured using a continuous state recorder (Y_t flow at time t). Randon *et al.* (2004) applied the Fuzzy Bayesian learning algorithm based on a Semi-Naive Bayes assumption. In this study the data were split into a training set consisting of 2090 examples from November 1972 to April 1974, and a test set of 1030 examples from November 1974 to December 1974. The objective of the experiment was to predict flow 36 hours in advance, corresponding to Y_{t+6}. It was assumed

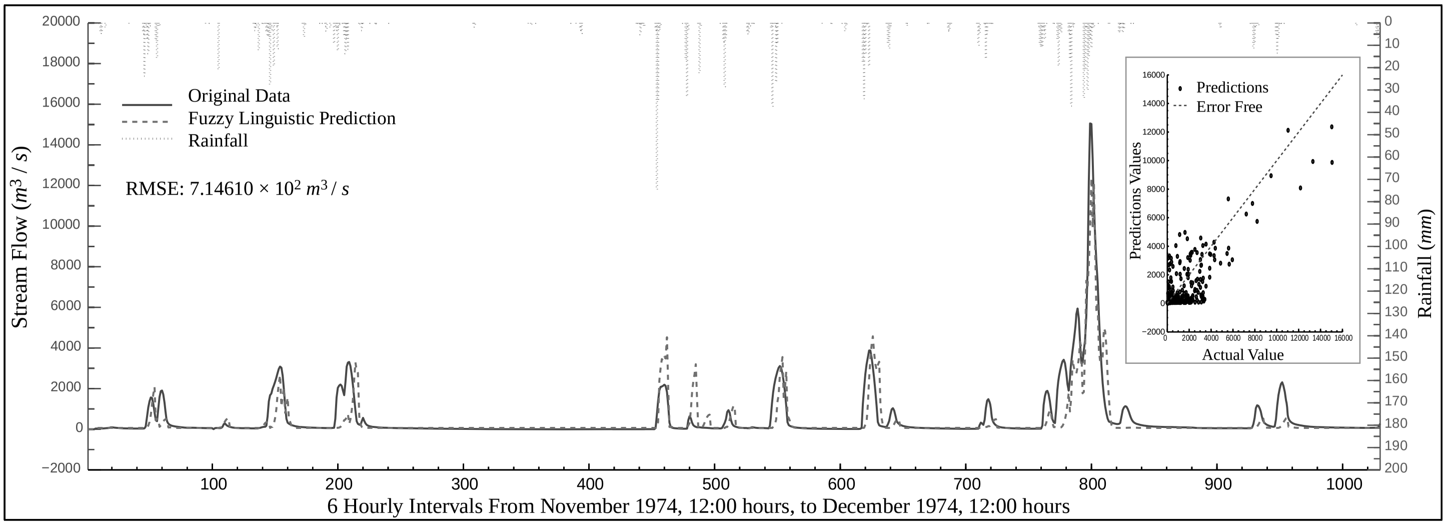


Fig. 8.7 Plots comparing actual flow against predicted flow applying the Fuzzy Bayesian prediction model (Randon *et al.* 2004). With kind permission from Springer Science & Business Media. (See the colour version of this figure in Colour Plate section.)

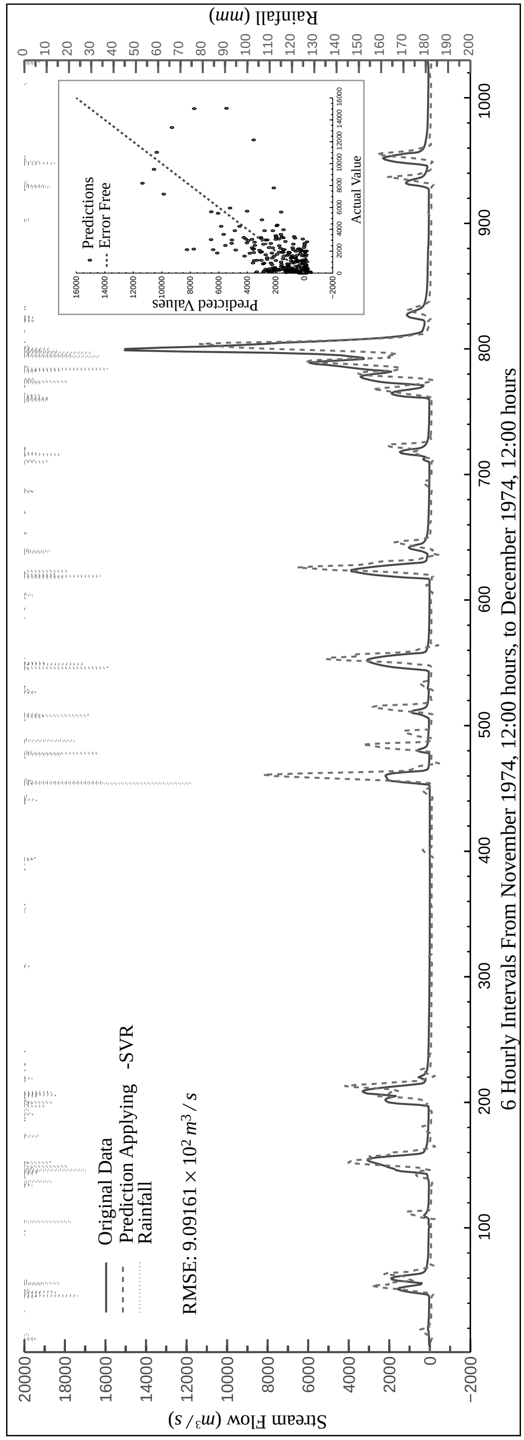


Fig. 8.8 Plots comparing actual flow against predicted flow applying an ε – SVR (Randon *et al.* 2004). (See the colour version of this figure in Colour Plate section.)

that another system was available to provide up-to-date rainfall measurements allowing values $U_{t+1}, \ldots, U_{t+5}$ to be included as features in the Fuzzy Bayesian Model. Other features included flow values $Y_{t-2}, \ldots, Y_t$ and rainfall measurements $U_{t-2}, \ldots, U_t$.

The application of the Fuzzy Bayesian algorithm results in a Root Mean Squared Error RMSE of $7.14610 \times 10^2\,\mathrm{m}^3\,\mathrm{s}^{-1}$ on the test dataset. Figure 8.8 shows the actual and predicted flow on this dataset as a time series plot, together with a scatter diagram of predicted against actual values. Figure 8.7 also shows, plotted on a different axis, the rainfall measurements for this time period. Randon *et al.* (2004) compared this performance with that of the regression support vector machine $\varepsilon - SVR$ (Vapnik 1995) (Fig. 8.8), using an implementation by Gunn (1998), and that of the transfer function based rainfall-runoff modelling system Weather Radar Information Processor WRIP (Cluckie and Han 2000); the RMSE for these methods was $9.09161 \times 10^2\,\mathrm{m}^3\,\mathrm{s}^{-1}$ and 9.47257×10^2 $\mathrm{m}^3\,\mathrm{s}^{-1}$, respectively.

For the WRIP system (Fig. 8.9) we can see that the streamflow has been modelled well. However, the results also show that WRIP gives a noisy prediction with large oscillations and some negative predicted values. This effect can also be seen, to a somewhat lesser extent, in the $\varepsilon - SVR$ results (Fig. 8.8). Furthermore, in both approaches while the low streamflow values are accurately predicted, the high values are underestimated. In contrast, the prediction accuracy of the Fuzzy Bayesian method is substantially better with a much lower RMSE than either WRIP or $\varepsilon - SVM$, and giving good estimates of both high and low flow values.

Randon *et al.* (2004) also reported on the application of the Fuzzy Bayesian model to evaluate the support for hypotheses involving fuzzy linguistic labels. For example, based on the labels *very low*, *low*, *high* and *very high*, Randon *et al.* (2004) evaluated a probability of 0.9994 for the following query:

What is the probability that if the rainfall U_{t+4} is *medium* and the rainfall U_{t+5} is *high*, with *high* streamflow values $Y_{t-1}, \ldots, Y_t$, then the streamflow Y_{t+6} is also *high*?

Lawry *et al.* (2004) introduced an algorithm for automatically extracting rules from a Fuzzy Bayesian model and applied this to the Bird Creek database. An example of an inferred rule using this system is as follows:

- IF U_{t+4} is not *very low* AND Y_{t-1} is only *high* AND U_{t+5} is *high* but not *low* AND Y_{t-2} is between *low* and *high* THEN Y_{t+6} is *high*: 0.991

Upper Severn catchment

The River Severn is situated in soutwest England with a catchment area that extends from the Cambrian Mountains in mid-Wales to the Bristol Channel in England. The Severn catchment can be seen in Fig. 8.11. Here we focus on the Upper Severn from Abermule in Powys and its tributaries, down to Buildwas in Shropshire. The River Severn dataset consists of 13,120 training examples from 1/1/1998 to 2/7/1999 and 2760 test examples from 8/9/2000 to 1/1/2001, recorded hourly. Each example has 19 continuous attributes. These fall into two categories: station level measurements and raingauge measurements. Both these measurements have significant uncertainty associated with them and raingauge measurements are particularly imprecise. Raingauges contain a significant amount of uncertainty in their calibration, and in particular in their tipping mechanism. In addition, data from raingauges are point measurements, and assume a uniform distribution of rainfall in the respective area. This spatial uncertainty is compounded due to the fact that raingauges can be positioned some distance (up to a few miles) from the riverbank. The output variable is the river level at Buildwas at time $t + \delta$.

McCulloch *et al.* (2008) applied ULID3, a real-time updateable version of LID3, to the Severn database. ULID3 allows for online updating of branch probabilities in linguistic decision trees, and for adapting the definition of labels in order to cover new values that fall outside the range of the original training data. For a lead time of $\delta = 24$ hours the predicted river level on the test data for

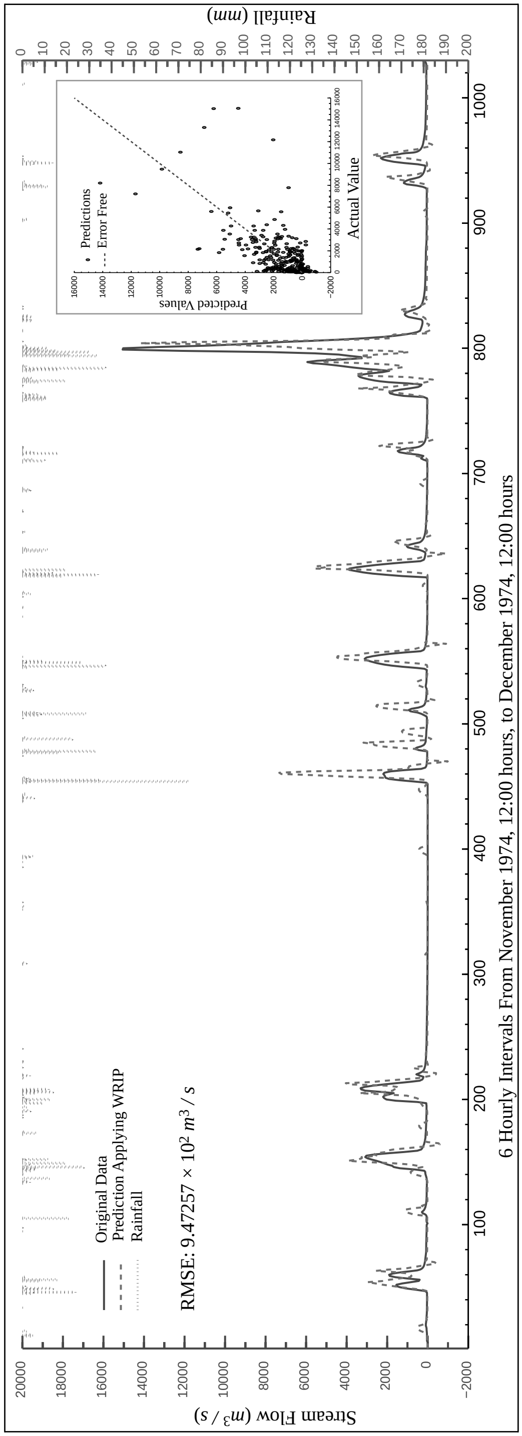
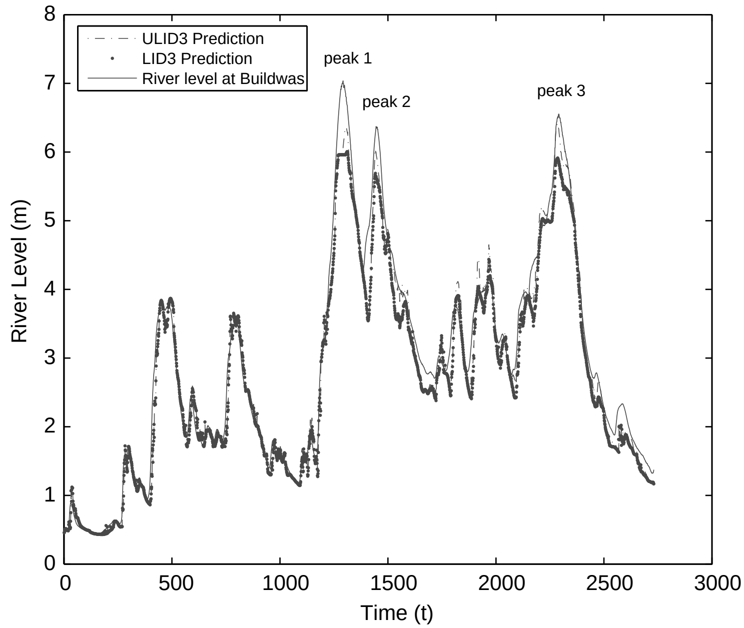
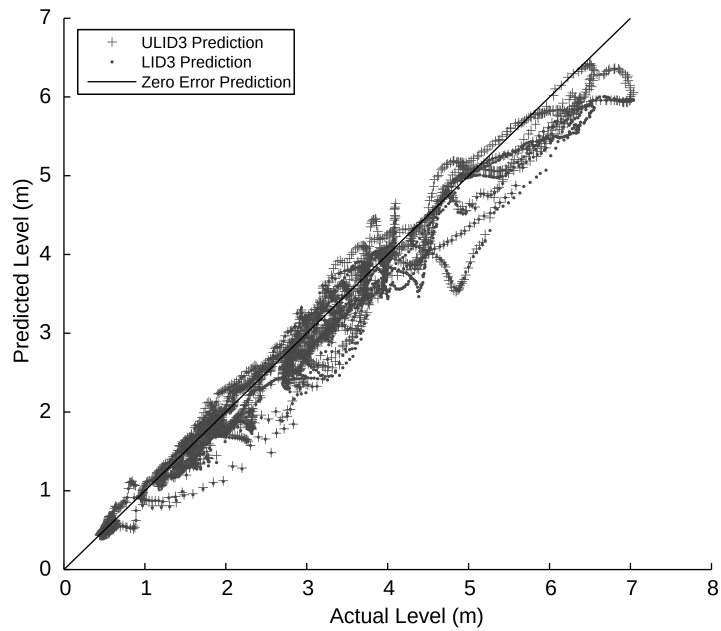


Fig. 8.9 Plots comparing actual flow against predicted flow applying the WRIP system. (See the colour version of this figure in Colour Plate section.)

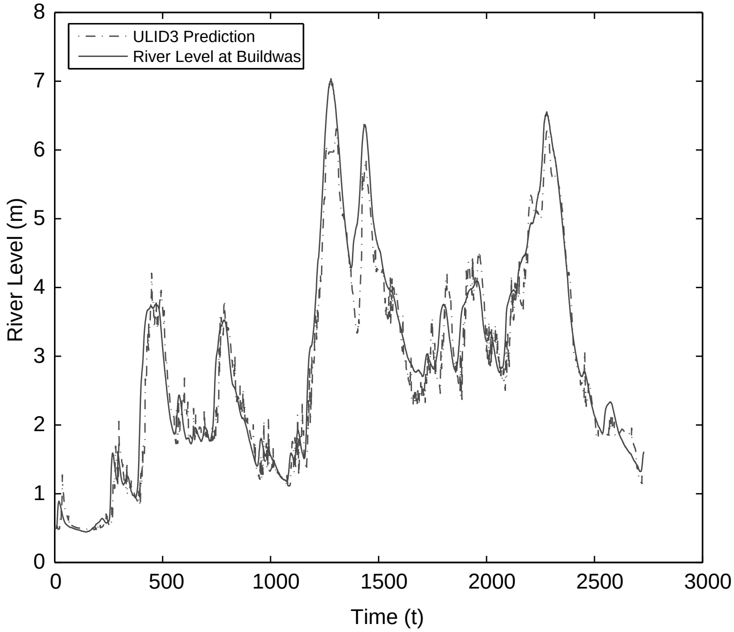


(a) Prediction of the River Level at Buildwas at $t + 24hrs$

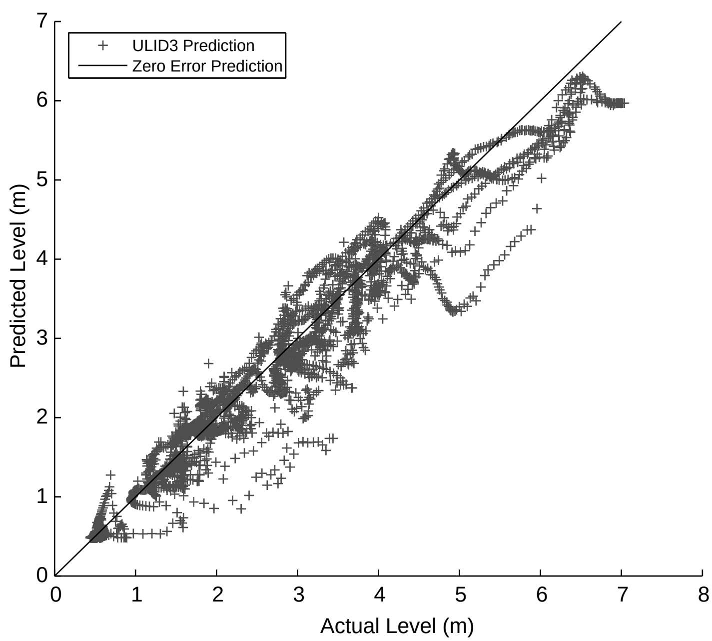


(b) Prediction of the River Level at Buildwas at $t + 24hrs$

Fig. 8.10 ULID3 predictions of the river level at Buildwas at 24 hours ahead. (a) Predicted river level on the test data for both LID3 and ULID3 shown as a time series, together with the actual measurement values. (b) Scatter plot of predicted against actual values for both algorithms. (See the colour version of this figure in Colour Plate section.)

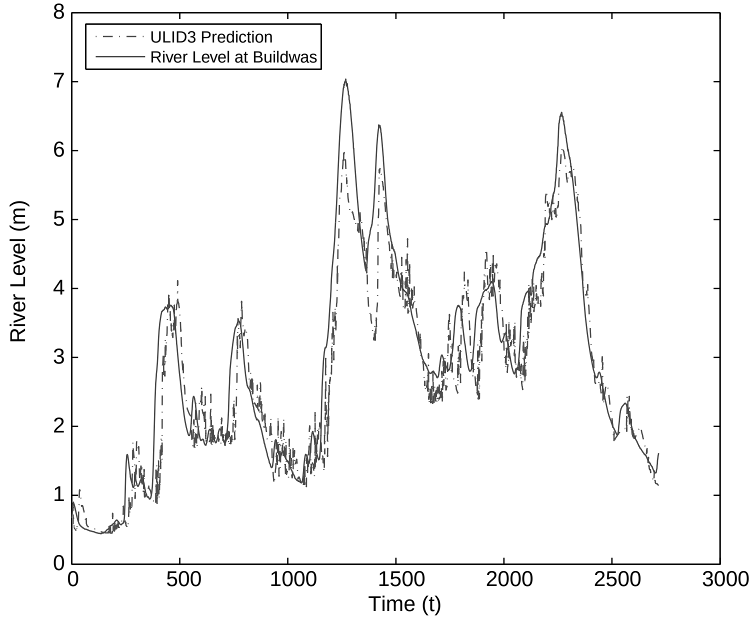


(a) Prediction of the River Level at Buildwas at t + 36 *hrs*

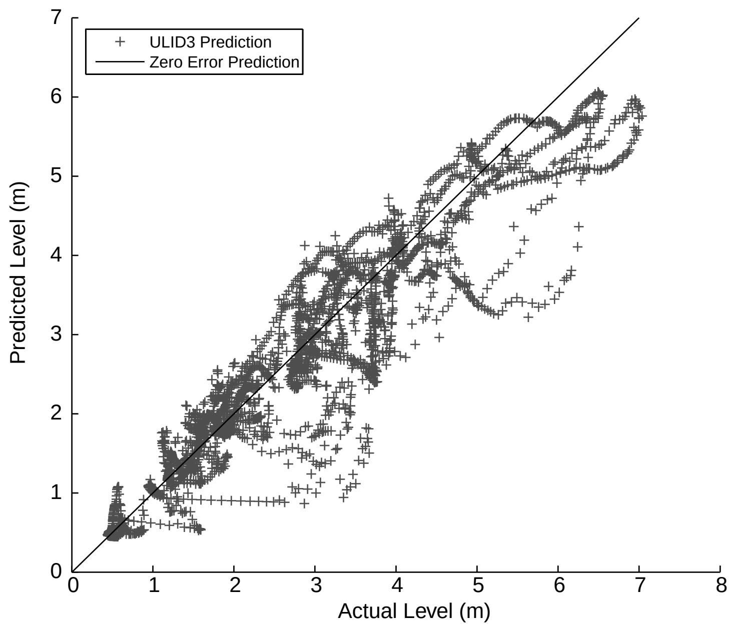


(b) Prediction of the River Level at Buildwas at t + 36 *hrs*

Fig. 8.11 ULID3 predictions of the river level at Buildwas at 36 hours ahead. (a) Predicted river level on the test data for ULID3 shown as a time series, together with the actual measurement values. (b) Scatter plot of predicted against actual values. (See the colour version of this figure in Colour Plate section.)



(a) Prediction of the River Level at Buildwas at $t + 48hrs$



(b) Prediction of the River Level at Buildwas at $t + 48hrs$

Fig. 8.12 ULID3 predictions of the river level at Buildwas at 48 hours ahead. (a) Predicted river level on the test data for ULID3 shown as a time series, together with the actual measurement values. (b) Scatter plot of predicted against actual values. (See the colour version of this figure in Colour Plate section.)

both LID3 and ULID3 is shown as a time series in Figure 8.10a together with the actual measurement values. Figure 8.10b shows a scatter plot of predicted against actual values for both algorithms. The three highest peaks shown in Figure 8.10a (labelled peak 1, peak 2 and peak 3) resulted in significant flooding and it is therefore crucial that any effective prediction algorithm should accurately predict these peak values with as long a lead time as possible.

All three peaks in the test data exceed the maximal output domain values present in the training data. These new data are therefore very hard to predict without updating the model. Consider peak 1, the highest peak shown in Figure 8.10a. LID3 is unable to interpolate beyond the range of values in the training data and consequently fails to predict the peak accurately. Figure 8.10a shows that ULID3 is more accurate than LID3 on the three peak values, although improvement on peak 1 is limited by the fact that since we are predicting 24 hours ahead there is a 24-hour delay before updating can occur. Further significant improvements are shown for peaks 2 and 3.

Figures 8.11 and 8.12 show the performance of ULID3 with lead times of $\delta = 36$ hours and $\delta = 48$ hours. Table 8.2 gives the RMSE for both LID3 and ULID3 for lead times ranging from $\delta = 24$ to $\delta = 72$. Although there is a notable difference in RMSE between $\delta = 24$ and $\delta = 36$, Figure 8.13a suggests that the performance on the three peaks is not significantly reduced. For $\delta = 48$ the error on the first peak is significantly increased although accuracy remains high for peaks 2 and 3 (see Fig. 8.12a). Overall, these results appear to be comparable with those obtained using the TF-Kalman filter model proposed by Romanowicz et al. (2006).

Table 8.2 Comparison of results with LID3 and ULID3 for prediction of river level at Buildwas at various lead times

	Accuracy (RMSE)	
$t + \delta$ hours	**LID3**	**ULID3**
$t + 72$ h	0.91	0.88
t + 66 h	0.86	0.82
$t + 60$ h	0.78	0.75
$t + 54$ h	0.70	0.67
$t + 48$ h	0.64	0.59
$t + 42$ h	0.54	0.50
$t + 36$ h	0.45	0.41
$t + 30$ h	0.38	0.33
$t + 24$ h	0.33	0.28

RMSE,.

Conclusions

In this chapter we have presented two AI learning algorithms within the Label Semantics framework. These can automatically generate fuzzy rule-based models from data and also incorporate probabilistic uncertainty. These methods have been shown to perform well across a number of case study problems in flood forecasting. Indeed, their predictive accuracy is comparable and sometimes better than that of more black-box approaches such as Neural Networks and Support Vector Machines. We have also given examples of rules extracted from the fuzzy models that give transparency to the forecasting process.

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9 Real-Time Updating in Flood Forecasting and Warning

PETER C. YOUNG

Introduction

A model-based flood warning system incorporates a catchment model of some sort, normally defined by a set of parameters (coefficients), together with a forecasting engine that utilizes this model to compute flow or level (stage) forecasts into the future, based on telemetered rainfall and flow data measured at various locations in the catchment area. Real-time updating involves the continual adaption of the model state variables, outputs and parameters, so that the forecasts for various times into the future are based on the latest available information and are optimized, in some sense, to minimize the forecasting errors. In the last few years, this process has been absorbed into the more comprehensive process of 'data assimilation' – namely a computer-controlled process where the data are assimilated into the computer system on a continuing basis and used to perform various tasks including state/parameter updating and flow forecasting/warning.

Since the monitored data are subject to noise contamination and uncertainty of various kinds, it is clear that the model should be defined in stochastic terms and the process of real-time updating should be considered from a statistical standpoint. The most obvious statistical method of implementing real-time updating is recursive estimation: see, for example, Gelb (1974), Ljung and Söderström (1983) and Young (1984). Here, the estimates of model parameters and state variables are updated at each sampling instant on the basis of the estimates obtained at the previous sampling instant.

Many different recursive algorithms for both parameter and state estimation have been proposed over the last 50 years: see, for example, the discussion on this topic in the books by Bryson and Ho (1969), Jazwinski (1970), Maybeck (1979), Ljung and Söderström (1983), Young (1984), Norton (1986), Harvey (1989), and Durbin and Koopmans (2001). In fact, recursive estimation dates back to the early 19th century, when Gauss first developed the Recursive Least Squares (RLS) algorithm sometime before 1826 [see Gauss (1826) and Appendix 2 of Young (1984), where the Gauss derivation of RLS is compared with the modern vector-matrix derivation]. RLS was rediscovered by Plackett (1950) and, 10 years later, Kalman developed his now famous recursive state estimation 'filter' (Kalman 1960), the core of which can be considered as a modified RLS algorithm for estimating the time variable states of a stochastic state space model.

Statistical methods of real-time state and parameter updating fall into two main groups: 'analytical' methods, such as the **Kalman Filter** (KF), which are derived analytically and can be solved simply using the resulting analytical expressions; and numerically intensive methods, normally based on **Monte Carlo Simulation** (MCS) analysis, where the statistical updates involve simple but computationally expensive ensemble averaging of some kind. Other examples of analytical recursive parameter estimation algorithms include: the recursive **Instrumental Variable** (IV) algorithm of Young (1970, 1974) and the related

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optimal **Refined Instrumental Variable** (RIV) algorithm, which is used in the illustrative example later in this chapter (Young 1976, 2008; Young and Jakeman 1979); the **Recursive Prediction Error** (RPE) algorithms of Ljung and Söderström (1983) and Stigter and Beck (2004); the **Bayesian Recursive Estimator** (BaRE) of Thiemann *et al.* (2001); and **DYNamic Identifiability Analysis** (DYNIA) proposed by Wagener *et al.* (2003).

Examples of recursive algorithms that include the use of MCS methods and exploit ensemble averaging are the **Ensemble Kalman Filter** (EnKF: e.g. Evensen 2007; Vrugt *et al.* 2005); the **Particle Filter** (PF: e.g. Gordon *et al.* 1993; Moradkhani *et al.* 2005a; Smith *et al.* 2006); and the **Unscented Kalman Filter** (UKF: e.g. Julier *et al.* 2000), all of which are discussed under 'Large and highly nonlinear stochastic systems' below. Note also that there are other 'Variational Data Assimilation' methods that have been used mainly in weather forecasting: these are not considered in this chapter but a comparison of the '4DVar' variational method and the EnKF, within a hydrological forecasting context, is available in Seo *et al.* (2003, 2009).

The KF is certainly the most famous recursive algorithm and it has received much attention in various areas of engineering, science and social science. Within the present context, its main limitation is that it only estimates and forecasts the **state** variables in the system, under the assumption that the parameters of the state space model are known exactly. As Kalman (1960) pointed out:

> . . . [it is] convenient to start with the model and regard the problem of obtaining the model itself as a separate question. To be sure, the two problems should be optimized jointly if possible; the author is not aware, however, of any study of the joint optimization problem.

This challenge was taken up quickly and several authors suggested approaches to recursive parameter estimation. Perhaps the best-known outcome of this research effort is the **Extended Kalman Filter** (EKF), first suggested by Kopp and Orford (1963). A good introduction to the EKF and related nonlinear, minimum variance, state estimation filters, including the iterated EKF, is given in Gelb (1974, p. 182 *et seq.*), and a tutorial example in a hydrological setting is given in Young (1984, p. 215 *et seq.*). In the EKF, parameter updating is carried out within a single, **nonlinear**, state space model, with the parameters considered as adjoined state variables described by simple state equations, such as the random walk (see 'Parameter updating' below); and this nonlinear model is then 'relinearized' at each recursive update.

The EKF is a simple, approximate solution to the optimal, nonlinear estimation and filtering problem, the general solution to which is infinite dimensional (Kushner 1967). Unfortunately, as an approximation, the EKF has various limitations, such as problems with covariance estimation and convergence for multi-dimensional, nonlinear models. As Julier *et al.* (2000) conclude:

> Although the EKF (in its many forms) is a widely used filtering strategy, over 30 years of experience with it. . . has led to a general consensus that it is difficult to implement, difficult to tune, and only reliable for systems that are almost linear on the time scale of the update intervals structure.

Not surprisingly, other approximate solutions are possible and some of these are discussed in the early but very influential book by Jazwinski (1970). However, such algorithms are normally based on higher order expansions and, while often theoretically superior, they do not possess the attractive, practical simplicity of the KF and EKF. It is for this reason that subsequent research in this area has taken the MCS-based route mentioned above. This is exemplified by the EnKF algorithm, which can be considered as a computationally more expensive but algorithmically simpler and more robust alternative to the EKF.

Within the specific flood forecasting context, Refsgaard (1997) gives a review of different updating techniques used in real-time flood forecasting systems, as well as the comparison of two different updating procedures applied to a conceptual hydrological model of a catchment, including rainfall-flow and the flow routing models. Unfortunately, the EKF algorithm that he considers has the limitations mentioned above. A recent, more comprehensive review of real-time forecasting is given in Chapter 5 of Beven (2009), which includes many more references than is possible in the present chapter.

In order to illustrate more clearly the nature of the parameter and state updating procedures, this mainly tutorial chapter utilizes a fairly simple, nonlinear rainfall-flow model as a practical example. However, these same procedures can be applied to any models if their state variables are observable[1] (see under 'Recursive state and parameter updating' below) and their parameters are clearly identifiable (see under 'The model and its parametric identifiability' below) from the available rainfall, flow and/or level data. Typical examples are **Hybrid-Metric-Conceptual** (HMC) and **Data-Based Mechanistic** (DBM) models (see below), which normally describe elemental rainfall-flow and flow-routing processes within the catchment and are most often of low dimension. At the catchment scale, however, these elemental models can be linked to produce quasi-distributed models of any size that is consistent with the availability of rainfall-flow or level data in the catchment. Moreover, recent developments in **emulation** modelling have shown how 'dominant mode' DBM and other low-order models are able to mimic the behaviour of large hydrodynamic simulation models and so form the basis for flood forecasting system design (see 'Dynamic emulation modelling' below).

This chapter does not address in any detail the real-time updating of large hydrodynamic and other distributed hydrological models, although it does outline procedures that currently have most promise in this context. The reason for this is that such models are rarely, if ever, statistically identifiable or observable from the data, so that real-time updating presents a variety of difficulties that depend on the nature of the specific model. Consequently, reasonably successful updating requires the imposition of very tight, model-specific, prior constraints and assumptions, not all of which can be tested rigorously. As a result, many problems remain to be solved (see the practical example described by Clark *et al*. 2008) before more general and systematic real-time updating procedures can be designed and recommended for this class of very large models.

[1] Note that this is a technical requirement on the state space model; it does not mean, of course, that they have to be directly observable by field measurement.

Catchment Models

Wheater *et al*. (1993) classify catchment models in a number of categories: Conceptual, Physics-Based, Metric and Hybrid-Metric-Conceptual (HMC). The models in the first two categories can be quite complex and can be either in the form of lumped parameter differential or difference equations; or distributed parameter partial differential (or difference) equations in time and space. The models in the latter two categories are normally much simpler and often consist of continuous or discrete-time, lumped parameter equations, where the metric variety (e.g. neural net, neuro-fuzzy, etc.) have a completely 'black-box' form; while the HMC variety have a mechanistic identity of some kind. Most HMC models are of the **hypothetic-deductive** kind, in which the model structure is assumed prior to model parameter estimation (optimization): typical examples are IHACRES (Jakeman *et al*. 1990), PDM (Moore 1985, 2007) and HYMOD (Moradkhani *et al*. 2005b). On the other hand, DBM modelling (e.g. Young and Lees 1993; Young 1993, 2001a, 2002; Young and Beven 1994; Young *et al*. 2004; Ratto *et al*. 2007b) is a particular type of **inductive** HMC modelling in which the mechanistic model structure is not limited by prior assumptions but inferred statistically from the data. This DBM approach is utilized later in this chapter (see 'Data assimilation and adaptive forecasting: an illustrative tutorial example') to illustrate how the real-time parameter and state updating procedures outlined in the next section are applied to a well-known set of rainfall-flow data.

In this digital age, all the models will be solved in a digital computer and so we will consider them within the following generic, discrete-time, stochastic structure:

$$\begin{aligned} \boldsymbol{x}_k &= F(\boldsymbol{x}_{k-1}, \boldsymbol{u}_k, \boldsymbol{\zeta}_k, \boldsymbol{\theta}) \\ \boldsymbol{y}_k &= G(\boldsymbol{x}_k, \boldsymbol{u}_k, \boldsymbol{\xi}_k) \\ \boldsymbol{v}_k &= H(\boldsymbol{u}_k, \boldsymbol{\eta}_k) \end{aligned} \tag{9.1}$$

where $\boldsymbol{x}_k$ is a state vector composed of state variables that are affected dynamically by an input vector $\boldsymbol{u}_k$ and a 'system' noise vector $\boldsymbol{\zeta}_k$ that normally represents unmeasured, stochastic disturbances to the system; $\boldsymbol{y}_k$ is an observation or 'output measurement' vector that is some combination of the state and input variables, as well as an output measurement noise vector $\boldsymbol{\xi}_k$; and $\boldsymbol{v}_k$ is a measurement of the input vector that is a function of the true input vector $\boldsymbol{u}_k$ contaminated in some manner by an input measurement noise vector $\boldsymbol{\eta}_k$. Finally, $\boldsymbol{\theta}$ is a $p \times 1$ vector of (normally unknown) parameters that define the inherent static and dynamic behaviour of the system. There may be other parameters that define the characteristics of the measurement equations but these have been omitted here both for simplicity and because it is often assumed that they will be known *a priori*.

Normally, the state variables $x_{1,k}$, $x_{2,k} \ldots x_{n,k}$ that comprise the state vector $\boldsymbol{x}_k$ of this n^{th} order, stochastic dynamic system will relate to the dynamic behaviour at specified spatial locations (in the present context, flow or level variations at stations along the river system) but, again for simplicity, the spatial index has been omitted here. If the model is of a distributed parameter form, then there will be many such spatial locations defined by whatever method of spatio-temporal discretization has been used to convert the partial difference equations to a discrete space-time form. In the case of a 'quasi-distributed', lumped parameter model, the locations will normally be far fewer and will be defined by those spatial locations that are most important (in the present context, the flow or level variations at various specified gauging stations along the river). And the simplest lumped parameter model will relate specified inputs to specified outputs.

In general, the system disturbances $\boldsymbol{\zeta}_k$ actually affect the states and are important in defining the underlying, 'real' behaviour of the dynamic system (in the present context, they are normally disturbances such as unmeasured input flows or losses to groundwater); while $\boldsymbol{\xi}_k$ and $\boldsymbol{\eta}_k$ are measurement or other noise sources that obscure our observation of this real dynamic behaviour. In this setting, the function of a model and forecasting system is to 'filter' off the measurement noise effects and reconstruct the true, underlying behaviour, including the stochastic input disturbance effects.

Of course, the model (Equation 9.1) has to be obtained in some manner, normally by reference to the measurement vectors:

$$\boldsymbol{y}_k = \left[y_{1,k}, y_{2,k} \ldots y_{p,k}\right]^T; \quad \boldsymbol{v}_k = \left[\boldsymbol{v}_{1,k}, \boldsymbol{v}_{2,k} \ldots \boldsymbol{v}_{m,k}\right]^T;$$
$$k = 1, 2, \ldots, N \tag{9.2}$$

where the superscript T denotes the vector transpose and N is the total sample size. In the present context, the elements of these vectors will normally take the form of telemetered measurements from remote, suitably located rainfall and flow/level gauges in the catchment. This will normally involve the specification or 'identification' of the detailed model structure and the 'estimation' or 'calibration' of the parameter vector $\boldsymbol{\theta}$ that characterizes this structure. Such an identification and estimation procedure presents a considerable challenge, and many different approaches have been suggested in the hydrological literature. These will not be reviewed in the present chapter since our object here is to consider how these model parameters and states can be updated online and in real-time **after** the model structure has been satisfactorily identified and a 'nominal' estimate $\hat{\boldsymbol{\theta}}_0$ of $\boldsymbol{\theta}$ has been obtained from prior, off-line studies.

One very important aspect of model parameter estimation is that the model being used as the basis for the forecasting system design must be 'identifiable' from the available data, in the sense that the model parameters must be well estimated from these data and there should be no ambiguity about their values. As discussed later, parametric identifiability requires that the model is not 'over-parameterized' or, if it is, that many of the parameters' values are assumed 'well known' prior to updating and only a small, and hopefully identifiable, subset have to be estimated. This latter approach is possible, either deterministically, by simply constraining the parameters to the assumed values; or stochastically, by imposing tight prior distributions on their values. Although such an approach is often used in practice, it is rather difficult to justify because it can imply an unreasonable level of trust in the prior assumptions. The situation can be improved, however, by performing sensitivity analysis (see, e.g., Ratto et al. 2007b; Saltelli *et al.* 2000) or by using some form of model 'emulation', as discussed later under 'Dynamic emulation modelling', where the large, over-parameterized model is emulated by a much smaller and identifiable 'dominant mode' model (Ratto *et al.* 2007a; Young and Ratto 2008).

Recursive State and Parameter Updating

Before proceeding to consider the general case of nonlinear, stochastic dynamic models, it is instructive to consider the special, but practically very useful, case of a linear dynamic model with stochastic input disturbances and measurement noise that have Gaussian normal amplitude distributions (or at least can be described sufficiently well by their first two statistical moments). As we shall see later, although this is a quite simple model, it has considerable practical relevance. In particular, it is quite common for rainfall-flow data to have simple nonlinear dynamic characteristics that allow for the utilization of linear forecasting methods, such as those discussed below.

Linear or near-linear stochastic systems

Normally, the generic, nonlinear, stochastic, dynamic model (Equation 9.1) can be simplified substantially for use in flood forecasting. The simplest form is, of course, the completely linear equivalent of this model with Gaussian normal stochastic disturbances. In the present context, however, it makes sense to allow the parameters in the model to vary in some manner over time, in order to allow for changes in the catchment dynamics or inadequacies in the model. The standard non-stationary, linear, state space model for a n^{th}-order **single input-single output** system, with no input noise, of the kind described later in the illustrative tutorial example, can be written in the following discrete-time form:

$$\begin{aligned} \boldsymbol{x}_k &= \mathbf{F}_k \boldsymbol{x}_{k-1} + \mathbf{G}_k u_{k-\delta-1} + \boldsymbol{\zeta}_k \quad & \boldsymbol{\zeta}_k &= N(\mathbf{0}, \mathbf{Q}_k) \\ y_k &= \mathbf{h}^T \boldsymbol{x}_k + g_i u_{k-\delta} + \xi_k & \xi_k &= N(0, \sigma_k^2) \end{aligned} \tag{9.3}$$

where $\mathbf{F}_k$ is a $n \times n$ state transition matrix; $\mathbf{G}_k$ is a $n \times 1$ input matrix; $\mathbf{h}$ is a $n \times 1$ observation vector; and g_i is a scalar gain that is present when the input has an instantaneous effect on the output variable y_k. In order that this model can be used for state updating and forecasting, it is necessary that the state variables in the vector $\boldsymbol{x}_k$ are **stochastically observable** (Bryson and Ho 1969) from the input and output measurements u_k and y_k. In simple terms, this means that the model is such that an optimal least squares estimate $\hat{\boldsymbol{x}}_k$ of $\boldsymbol{x}_k$ (see below) can be computed from these measurements and that the variance of the state estimation error $\boldsymbol{x}_k - \hat{\boldsymbol{x}}_k$ is reduced by these operations. Note that observability does not require that the state variables are measured directly; it requires only that the structure of the model is such that the unambiguous **estimates** of the state variables can be reconstructed from the measured input and output variables u_k and y_k.

The subscript k on $\mathbf{F}_k$ and $\mathbf{G}_k$ is introduced to allow the elements of these matrices (the model parameters) to change with time. However, because the linear relationship between the scalar flow measurement and the state variables is normally time-invariant, the observation vector $\mathbf{h}$ is not made a function of the sampling index k. Also, it is likely that the nominal model will be estimated off-line before this is utilized on-line, in real time. The state space matrices for this nominal model will be represented by $\mathbf{F}_0$ and $\mathbf{G}_0$.

As shown, the $n \times 1$ vector of stochastic disturbances $\boldsymbol{\zeta}_k$ is assumed to have a multivariate normal distribution, with zero mean value and covariance matrix $\mathbf{Q}_k$; while the scalar measurement noise ξ_k is assumed to have zero mean and variance σ_k^2, which is made a function of the sampling index k in order to allow for heteroscedasticity in the observational errors. In general, $\mathbf{Q}_k$ is also assumed to change over time; as we shall see, however, in the practical example considered later, it is assumed that $\mathbf{Q}_k = \mathbf{Q}\,\forall\, k$.

The Kalman Filter Forecasting Algorithm

The linear, Gaussian, state space model (Equation 9.3) discussed in the previous section provides the basis for the implementation of the KF state estimation and forecasting algorithm. This algorithm has become very famous since Kalman published his seminal paper in 1960, with reported applications in almost all areas of science, engineering and social science. It is unlikely, however, that many users have actually read this paper. For example, while the KF can be usefully interpreted in Bayesian estimation terms and, indeed, seems the very embodiment of Bayesian estimation, it was actually derived using orthogonal projection theory and so does not rely upon Gaussian assumptions. For this and other reasons, it has proven to be a very robust algorithm that is ideal for practical applications. The KF algorithm is best written (Young 1984) in the following prediction-correction form, under the assumption that the system (Equation 9.3) is stochastically observable:

***A priori* prediction:**

$$\hat{\boldsymbol{x}}_{k|k-1} = \mathbf{F}_k \hat{\boldsymbol{x}}_{k-1} + \mathbf{G}_k u_{k-\delta-1} \tag{9.4a}$$

$$\mathbf{P}_{k|k-1} = \mathbf{F}_k \mathbf{P}_{k-1} \mathbf{F}_k^T + \mathbf{Q}_k \tag{9.4b}$$

***A posteriori* correction:**

$$\hat{\boldsymbol{x}}_k = \hat{\boldsymbol{x}}_{k|k-1} + \mathbf{P}_{k|k-1}\mathbf{h}\left[\sigma_k^2 + \mathbf{h}^T \mathbf{P}_{k|k-1}\mathbf{h}\right]^{-1} \left\{y_k - \mathbf{h}^T \hat{\boldsymbol{x}}_{k|k-1-g_i u_{k-\delta}}\right\} \tag{9.4c}$$

$$\mathbf{P}_k = \mathbf{P}_{k|k-1} - \mathbf{P}_{k|k-1}\mathbf{h}\left[\sigma_k^2 + \mathbf{h}^T \mathbf{P}_{k|k-1}\mathbf{h}\right]^{-1} \mathbf{h}^T \mathbf{P}_{k|k-1} \tag{9.4d}$$

In theory, because the model is linear and the stochastic disturbances $\boldsymbol{\zeta}_k$ and ξ_k are assumed to be normally distributed random variables, the error on the estimate of the state $\hat{\boldsymbol{x}}_k$ is also normally distributed and, in the above KF equations, $\mathbf{P}_k$ is the error covariance matrix associated with $\hat{\boldsymbol{x}}_k$. The subscript notation $k|k-1$ denotes the estimate at the k^{th} sampling instant (normally an hour or a day in the present hydrological context), based on the estimate at the previous $(k-1)^{th}$ sampling instant.

The estimate $\hat{y}_k$ of the output variable y_k is obtained as the following linear function of the state estimates and any instantaneous input effect, by reference to the observation equation (Equation 9.3):

$$\hat{y}_k = \mathbf{h}^T \hat{\boldsymbol{x}}_k + g_i u_{k-\delta} \tag{9.4e}$$

The f-step-ahead forecasts of the output variable $\hat{y}_{k+f|k}$ are obtained by simply repeating the prediction f times, without correction (since no new data over this interval are available). It is straightforward to show that the f-step-ahead forecast variance is then given by:

$$\mathrm{var}\left(\hat{y}_{k+f|k}\right) = \hat{\sigma}_k^2 + \mathbf{h}^T \mathbf{P}_{k+f|k}\mathbf{h} \tag{9.4f}$$

where $\mathbf{P}_{k+f|k}$ is the error covariance matrix estimate associated with the f-step-ahead prediction of the state estimates. This estimate of the f-step-ahead prediction variance is used to derive approximate 95% confidence bounds for the forecasts, under the approximating assumption that the prediction error can be characterized as a nonstationary Gaussian process (i.e. twice the square root of the variance at each time step is used to define the 95% confidence region). The derivation of the above KF equations is not difficult but outside the scope of this chapter. However, the interested reader can find this derivation in any tutorial text on estimation theory, such as Maybeck (1979), Young (1984) or Norton (1986). The derivation is also sketched out in the FRMRC 'User Focussed Measurable Outcome' Report UR5 (Young *et al.* 2006).

State and output updating

In the above KF equations (Equations 9.4a and 9.4b), the model parameters that define the state space matrices $\mathbf{F}_k$ and $\mathbf{G}_k$ in Equation 9.3 are known initially from the model identification and estimation analysis based on an estimation dataset. However, by embedding the model equations within the KF algorithm, we have introduced additional, unknown parameters, normally termed 'hyper-parameters' to differentiate them from the model parameters.[2] In this example, these hyper-parameters are the variance σ_k^2 and the elements of the covariance matrix $\mathbf{Q}_k$. In practical terms, it is normally sufficient to assume that $\mathbf{Q}_k$ is purely diagonal in form, where the diagonal elements specify the nature of the stochastic inputs to the state equations and so define the level of uncertainty in the evolution of each state (in the later illustrative tutorial example, the 'quick' and 'slow' water flow states). The inherent state adaption of the KF arises from the presence of the $\mathbf{Q}_k$ hyper-parameters, since these allow the estimates of the state variables to be adjusted to allow for presence and effect of the unmeasured stochastic disturbances that naturally affect any real system.

Clearly, the hyper-parameters have to be estimated in some manner on the basis of the data. One well-known approach is to exploit **Maximum Likelihood** (ML) estimation based on **Prediction Error Decomposition** (see Schweppe 1965, 1973; Young 1999b). Another, used in the quite complex, multi-input, multi-site catchment network of the River Severn forecasting system (Romanowicz *et al*. 2006), is to optimize the hyper-parameters by minimizing the variance of the multi-step-ahead forecasting errors. In effect, this optimizes the memory of the recursive estimation and forecasting algorithm (see, e.g., Young and Pedregal 1999) in relation to the rainfall-flow or water level data. The main advantage of this latter approach is, of course, that the integrated model-forecasting algorithm is optimized directly in relation to the main objective of the forecasting system design – namely the minimization of the multi-step prediction errors. In the illustrative and much simpler single-input, single-site example given later in this chapter, however, the hyper-parameters are 'tuned' manually in order to better explain their effect on the forecasts.

[2] Of course this differentiation is rather arbitrary since the model is inherently stochastic and so these parameters are simply additional parameters introduced to define the stochastic inputs to the model when it is formulated in this state space form.

Parameter updating

Within a hydrological context, **Time-Variable Parameter** (TVP) parameter estimation methods have been implemented recursively in two main ways: first, by joint state-parameter estimation using either state augmentation (as in the EKF), or by the use of parallel, interactive filters (e.g. Todini 1978; Liu and Gupta 2007); and second, by estimating the parameters separately from the state estimation (e.g. Young 1984, 1999a, 1999b, 2002; Romanowicz *et al*. 2006; Lin and Beck 2007).

Here, we will consider only the second category, where the TVPs are estimated separately to the state variables. This decision is based on two factors: first, separate estimation provides a more flexible approach to real-time updating; and second, it is supported by conclusions reached in one of the most recent of the above publications (Lin and Beck 2007). Beck and his co-workers have previously favoured a specific, improved formulation of the EKF approach, but they have now developed a powerful **Recursive Prediction Error** (RPE) algorithm inspired by the ideas of Ljung (1979), who carried out early research aimed at overcoming some of the 'notorious difficulties of working with the EKF as a parameter estimator'. On the basis of experience, over many years, Lin and Beck (2007) conclude that 'as a parameter estimator, the RPE algorithm has many advantages over the conventional EKF'.

The RPE approach of Lin and Beck has some similarity with an alternative method of separate TVP estimation, the **Refined Instrumental Variable** (RIV) algorithm (see, e.g., Young 1984, 2008). Both are optimal maximum likelihood estimation algorithms but are not limited in this sense; both can be applied to both discrete-time and continuous-time models; and finally, both model the parameters by a stochastic **Generalized Random Walk** (GRW) process. Indeed, the RPE and RIV algorithms are quite complementary so that, taken together, they constitute a powerful general approach to TVP estimation. However, the RPE algorithm is formulated within a 'hypothetico-deductive' framework, where the model structure is postulated on the basis of assumptions regarding the physical nature of the system and then the constant or time variable parameters that characterize this structure are estimated from the available time-series data.

On the other hand, the RIV algorithm is formulated within an 'inductive' framework, where no prior assumptions about the model structure are made, other than that the system can be modelled by a fairly general set of nonlinear differential equations or a discrete-time equivalent of this, such as Equation 9.3. In this inductive approach, the interpretation of the model in physically meaningful terms **follows** the parameter estimation phase, which may include both TVP and the related **State-Dependent Parameter** (SDP) estimation methods (Young *et al.* 2001; Young and Ratto 2008).

The recursive form of both the RPE and RIV algorithms is based on the assumption that the parameter vector $\boldsymbol{\theta}_k$ evolves over time as one of the GRW family of stochastic processes (Jakeman and Young 1984). The simplest example of this family is the following **Random Walk** (RW) model, which derives from the earliest research on TVP estimation (Lee 1964) and is used later in the illustrative example:

$$\boldsymbol{\theta}_k = \boldsymbol{\theta}_{k-1} + \boldsymbol{\eta}_k \quad \boldsymbol{\eta}_k = N(0, \mathbf{Q}_{\boldsymbol{\theta}}) \tag{9.5}$$

Both estimation algorithms then take the general recursive form:

$$\hat{\boldsymbol{\theta}}_k = \hat{\boldsymbol{\theta}}_{k-1} + \Pi_k \left\{ y_k - \hat{y}_k\left(\hat{\boldsymbol{\theta}}_{k-1}\right)\right\} \tag{9.6}$$

where, in the present context, $\hat{y}_k(\hat{\boldsymbol{\theta}}_{k-1})$ is the estimated flow generated by the model incorporating the estimated parameter vector $\hat{\boldsymbol{\theta}}_{k-1}$ obtained at the previous recursion, so that $y_k - \hat{y}_k(\hat{\boldsymbol{\theta}}_{k-1})$ constitutes the current 'innovations' error. The time variable gain matrix Π_k is a function of the parameter estimation error covariance matrix $\mathbf{P}_{\boldsymbol{\theta},k}$, which is also updated recursively. This recursive update includes the addition of the covariance matrix $\mathbf{Q}_{\boldsymbol{\theta}}$, which signals the possibility of change in the model parameters and allows this change to be estimated in order to reduce the innovation error and, in consequence, the response error between the model output and the measured output. In order to apply this algorithm, it is necessary to specify the initial *a priori* estimates of the model parameter vector and its associated covariance matrix, i.e. $\hat{\boldsymbol{\theta}}_0$ and $\mathbf{P}_{\boldsymbol{\theta},0}$, respectively.

Although the $\mathbf{Q}_{\boldsymbol{\theta}}$ matrix could have different diagonal elements, optimized by maximum likelihood (Young 1999b),[3] it is often restricted to a diagonal form $\mathbf{Q}_{\boldsymbol{\theta}} = \delta_\theta \mathbf{I}_p$, where $\mathbf{I}_p$ is the p^{th} order identity matrix and δ_θ is an associated scalar hyperparameter that can be optimized or 'tuned' to provide good tracking performance. However, the details of this and other aspects of the recursive RIV and RPE algorithms are given in the above cited references and will not be considered further here.

The RPE algorithm is fairly new but it seems to be quite robust in practical application, although a customized version of the algorithm would have to be written for any specified state space model. The RIV algorithm provides a general tool for transfer function estimation that does not require customization, and it has been used successfully in numerous practical applications for many years (see references in previous section), where its instrumental variable nature makes it particularly robust to the kind of stochastic disturbances and noise encountered in hydrological data. Discrete and continuous-time (RIVC) versions of the RIV algorithm are available as the *riv*, *rivbj*, *rivc* and *rivcbj* routines in the CAPTAIN Toolbox for Matlab.[4]

[3] In fact, the normal formulation involves the optimization of a normalized 'noise-variance ratio' matrix: see the cited references.

[4] This can be downloaded from http://www.es.lancs.ac.uk/cres/captain/ and has been used for all of the analysis reported in this chapter.

Hybrid continuous-discrete time updating

Although the previous subsections have assumed that the model is formulated in discrete-time, continuous-time models can be handled in a similar manner. As regards state updating, the most obvious approach is to utilize the continuous-discrete time form of the KF (see, e.g., Young 1984, p. 215 *et seq*.). Here, the model prediction step is carried out in continuous-time, normally by the numerical integration of the continuous-time model equations; while the correction step, which is likely to involve only discrete-time sampled data, is retained in the same form as Equations 9.4c and 9.4d. This formulation has the additional advantage of allowing for irregularly sampled data.

Large and highly nonlinear stochastic systems

As we shall see later, normally the nonlinear rainfall-flow model can be decomposed into a serial connection of an 'effective rainfall' input nonlinearity feeding into a purely linear system that defines the unit hydrograph properties of the system (termed a 'Hammerstein' model in the Systems literature). In this situation, the linear methods of updating outlined in the previous subsection can be amended simply for use in nonlinear rainfall-flow modelling and flow forecasting. However, if the nonlinearity is internal to the model and the model is complex, then it is necessary to consider more general methods that are not as restricted as the modified linear procedures. Here, we will consider briefly two such methods that have been utilized in a hydrological context and are relevant to the illustrative practical example described later: the EnKF and the PF. These have been compared recently by Weerts and Serafy (2006) when applied to the conceptual rainfall-runoff model HBV-96 for flood forecasting purposes. The related and rather quaintly named **Unscented Kalman Filter** (UKF) method is also discussed briefly.

The Ensemble Kalman Filter (EnKF)

The EnKF is an adaptation of the standard, analytic KF algorithm to nonlinear systems using Monte Carlo sampling and ensemble averaging in the prediction step and linear updating in the correction step. A definitive account of the EnKF appeared with the publication of Evensen's book on the subject (Evensen 2007), and the recent paper by Clark *et al*. (2008) provides a comprehensive and critical evaluation of both the EnKF and the related **Ensemble Square Root Filter** (EnSRF) when applied to the distributed hydrological model TopNet of the Wairau River basin in New Zealand. There are various ways in which joint state/TVP estimation can be carried out within an EnKF framework, but a relevant one in the present context is that suggested by Moradkhani *et al*. (2005b) and tested on data from the Leaf River in the USA (the same data as those used for the example given in subsequent sections of the present chapter).

The basic implementation of the EnKF for state estimation and forecasting is quite simple because the correction step in the recursions is the same as the standard KF. The Monte Carlo sampling and ensemble averaging is only required in the prediction step, which simply involves the computation of the ensemble mean and its associated covariance matrix, computed from the deviations of the ensemble members from the mean (acting as a surrogate for the true state, which is unknown, of course). However, Moradkhani *et al*. develop a 'dual EnKF', which requires separate state-space representation for the state variables and parameters through two linked algorithms (filters) running in parallel. Here, the parameters are treated in a similar manner to the state variables, with the parameters assumed to follow a stochastic RW process, exactly the same as that used for the implementation of recursive TVP estimation discussed above. However, the implementation of the dual recursions could be accomplished in various ways, and this is not all that clear from the description in the paper.

The EnKF results obtained with the Leaf River data are promising but they suggest the need for further research on the practical implications of the filter in relation to real-time state/parameter updating. For example, the flow forecasts appear good but it may be that these are estimates rather than one-day-ahead forecasts (see 'Comments' following the illustrative example given below).

Also the parameter estimation convergence is fairly slow when compared to that of the recursive RIV estimation algorithm applied to the same data (see later Fig. 9.12) and so some questions remain about how useful an EnKF implementation, such as this, would be in tracking time-variable parameters. Finally, the HYMOD model used in the study is quite small and simple, so it is not clear how well this EnKF approach would work in the case of a large, highly nonlinear model for which the EnKF is really intended (see Weerts and Serafy 2006; Clark *et al.* 2008).

The particle filter (PF)

When interpreted in Bayesian terms, the KF can be considered as a very special, analytically tractable version of the general recursive Bayesian Filter, as obtained when the state space model and observation equations are linear and the additive stochastic disturbance inputs have Gaussian amplitude distributions. The PF, on the other hand, is a sequential, Monte Carlo-based approximate mechanization of the prediction and correction stages of a fairly general recursive Bayesian filter (Gordon *et al.* 2000; Doucet *et al.* 2001; Moradkhani *et al.* 2005a; Smith *et al.* 2006) and so it applies to general nonlinear models with nonlinear observations and non-Gaussian stochastic disturbances. In the PF, the underlying posterior probability distribution function is represented by a cloud of particles in the state space and the samples automatically migrate to regions of high posterior probability. Moreover, in theoretical terms, convergence is not particularly sensitive to the size of the state space.

On the basis of this description, the PF seems extremely flexible and potentially very attractive. As so often with general methods such as this, however, there are practical drawbacks. It is naturally very expensive in computational terms, and practical restrictions on the number of particles that can be used in sampling the prior distributions often lead to posterior distributions that are dominated by only a few particles. This can introduce a need for modifications, such as the use of techniques that include residual resampling, Markov chain Monte Carlo (MCMC) analysis, sequential importance sampling and sampling importance resampling: see, for example, Moradkhani *et al.* (2005a). The latter reference and the one by Smith *et al.* (2006) are interesting because they also analyze the Leaf River data, although only in an estimation, rather than a forecasting sense.

The results obtained by Moradkhani *et al.* are rather mixed: for example, some of the high flows fall outside of the estimated uncertainty intervals and there is high interaction between the estimated states and parameters, to the detriment of the parameter estimates (also a common characteristic of the EKF and perhaps an argument, once again, for the separation of state and parameter estimation). Similar comments to those made about the EnKF in the last subsection apply to the PF: namely, (i) it is not clear how well the PF approach would work in the case of a large, highly nonlinear model rather than the simple HYMOD model used by Moradkhani *et al.*; and (ii) the parameter estimation results show fairly slow convergence and its time variable parameter tracking ability is questionable.

On the other hand, the results obtained by Smith *et al.* make good sense and they allow the authors to investigate the shortcomings in the HYMOD model structure. Also, the time variable parameter tracking results seem quite reasonable, although some of the estimated variations are rather volatile when compared with the recursive parameter estimates obtained in the example given later in this chapter, using the same data, and a similar complexity model.

Finally as regards the comparison of the EnKF and PF, Weerts and Serafy (2006) (see earlier) conclude that: 'For low flows, [the] EnKF outperforms both particle filters [the *Sequential Importance Resampling* (SIR) filter; and *Residual Resampling* filter (RR) variations], because it is less sensitive to mis-specification of the model and uncertainties.'

The unscented Kalman filter (UKF)

The UKF operates on the premise that it is easier to approximate a Gaussian distribution than it is to approximate an arbitrary nonlinear function (see, e.g., Julier *et al*. 2000, and the prior references therein). Instead of linearizing using Jacobian matrices, as in the EKF, the UKF uses a **deterministic** 'sigma point filter' sampling approach to capture the mean and covariance estimates with a minimal set of sample points. It has some similarities with the EnKF but the random sampling strategy of the EnKF is replaced by this deterministic approach, which will be more efficient both computationally and statistically when its assumptions are satisfied. The UKF appears to be a powerful nonlinear estimation technique and has been shown to be a superior alternative to the EKF in a variety of applications including state estimation and parameter estimation for time series modelling. So it clearly has potential relevance to rainfall-flow modelling and forecasting, although it is not clear whether it has been evaluated yet in this context. However, a dual-UKF method involving state and parameter estimation is described by Tian *et al*. (2008) in relation to the design of a system for assimilating satellite observations of soil moisture using the NCAR Community Land Model.

Dynamic Emulation Modelling

One new approach to real-time updating in the case of large and complex system models is the development of a Dynamic Emulation Model (DEM). Here, a large, and normally over-parameterized dynamic simulation model is emulated by a much smaller and identifiable 'dominant mode' model, such as a DBM model (Ratto *et al*. 2007a; Young and Ratto 2008). The process of emulation is shown diagrammatically in Figure 9.1. The large simulation model is first subjected to planned dynamic experiments and the data so obtained are used to identify and estimate the low order 'dominant mode' model (Young 1999a). For instance, a flood routing model such as ISIS or HEC-RAS (Hydrologic Engineering Center River Analysis System) is run in an unsteady flow mode and forced with an upstream boundary condition defined by specified flow inputs. The water surface level field generated by the unsteady simulation run is then used as a dataset for identification and estimation of a 'nominal' DEM, using an estimation algorithm, such as the RIV algorithm introduced earlier, or nonlinear State-Dependent Parameter versions of this (Young 2001b). This process is repeated for selected values of the large model parameters, and the relationship between these parameters and the DEM model parameters is inferred, using some suitable mapping method, so as to produce the complete DEM that behaves like the large model over the whole range of selected model values.

In order to validate the model, further simulations are then carried out and the ability of the DEM to emulate the large model behaviour under these changed circumstance is evaluated. A typical example of DEM validation is shown in Figure 9.2, which is taken from recent papers by Beven *et al*. (2008) and Young *et al*. (2009) and concerns the emulation of the HEC-RAS model. The DEM is in the form of a simple, nonlinear, dominant-mode DBM model identified and estimated from dynamic experimental data of the kind mentioned above. The output of the DEM is compared with HEC-RAS model validation data based on its response to a new set of upstream level data. As we see, the DEM outputs are very similar to the HEC-RAS model outputs at the six sites selected for emulation.

If the DEM is able to emulate the large model well, then it is clear that it can replace it for a variety of purposes, such as data assimilation, forecasting, automatic control and sensitivity analysis, for which the concept of emulation modelling was originally conceived: see, for example, Conti *et al*. (2007). Moreover, because it is a low-order, well-parameterized and identifiable model, it can be updated in real time using recursive estimation algorithms such as RIV and RPE. Although research on dynamic emulation modelling is at an early stage, the initial results are promising and it clearly represents a potential approach to catchment modelling that allows for fairly straightforward real-time updating.

The Model and Its Parametric Identifiability

In the previous sections, it has been assumed that the catchment model used for forecasting is

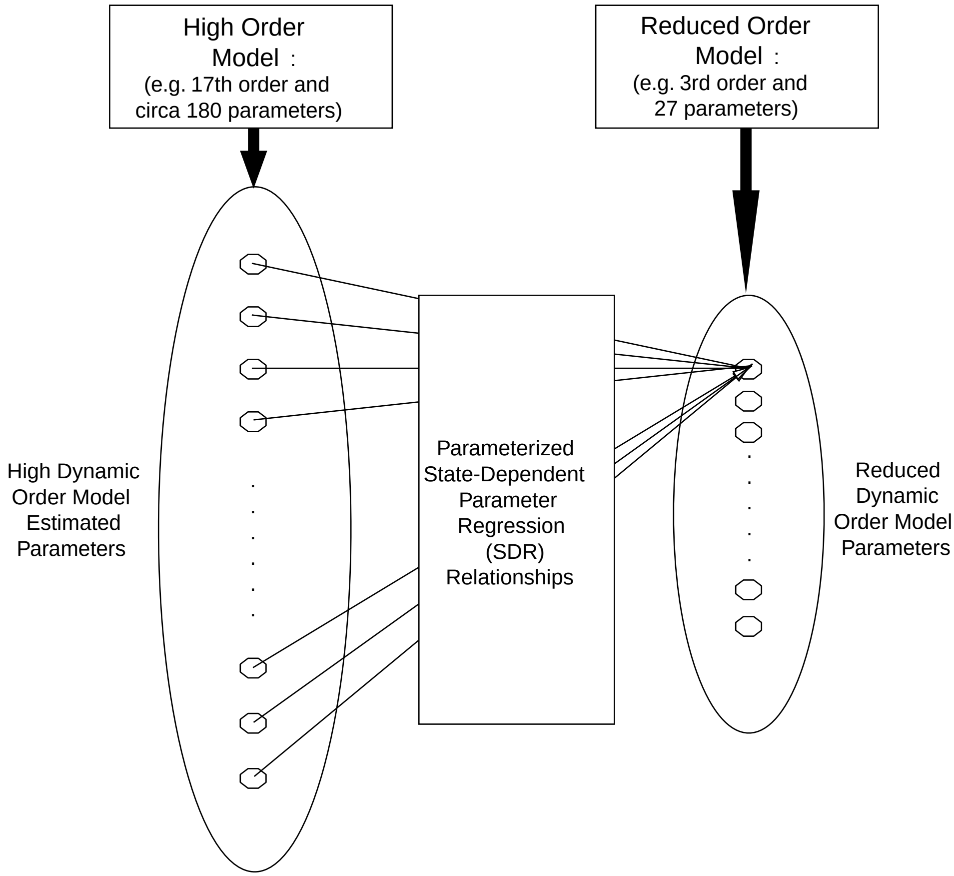

Fig. 9.1 The process of dynamic emulation model synthesis.

parametrically identifiable from the available data. In simple terms, a model is identifiable from data if there is no ambiguity in the estimates of its parameters. In relation to input-output models of the kind used in catchment modelling, identifiability is affected by the following interrelated factors.

1 The nature of the input signal: It can be shown that, ideally, the input signal should be 'persistently exciting' in the sense that it remains bounded in mean and variance, while continuing to perturb the system sufficiently to allow for unambiguous estimation of the model parameters: see, for example, Young (1984). This requirement is linked strongly with the dynamic order of the system:[5] for instance, in the case of linear systems, it can be shown that a single pure sinusoid will only provide sufficient excitation to identify a system described by a first-order differential or difference equation; in general, the input should contain at least n sinusoids of distinct frequencies to allow for the identification of an n^{th}-order system. In the case of rainfall-flow or level data, although the rainfall may not be

[5] In a hydrological system, the dynamic order is linked to the number of 'tanks' or 'reservoirs': so a typical first-order model would be a first-order differential equation of single river reach or storage reservoir; and a second-order model would be two such reservoirs connected in series, parallel or feedback.

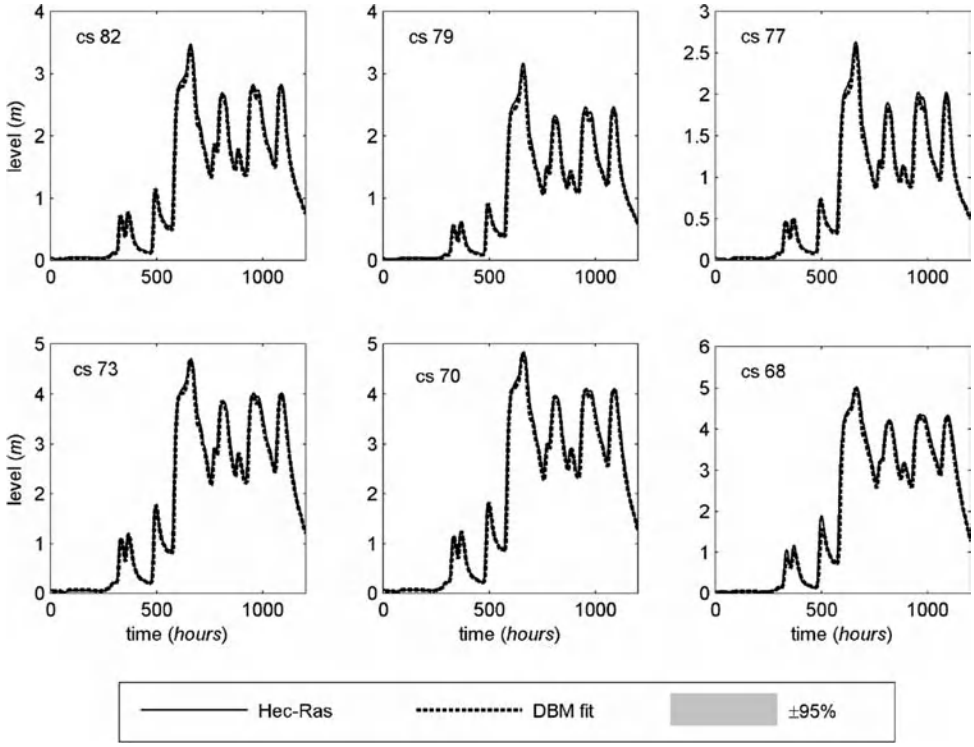

Fig. 9.2 Validation of a Dynamic Emulation Model (DEM) for the HEC-RAS (Hydrologic Engineering Center River Analysis System) model at six downstream sites based on a new set of upstream level data.

persistently exciting in this complete theoretical sense, it will normally provide sufficient excitation to ensure identifiability (except for the case of summer inactivity in ephemeral catchments). The situation is less clear in the case of flow-flow or level-level routing, since the upstream inputs may well be very smooth and relatively 'unexciting' in some cases.

2 The number of parameters in the model: In a lumped parameter model, the number of parameters is defined by the dynamic order and structure of the system, as well as any additional parameters that, for example, are associated with the characterization of nonlinear functions appearing in the model equations. However, the most important parametric contribution is concerned with defining the order of the system, since it is this that affects identifiability most critically and links in with item 1 above. In the case of distributed parameter partial differential equations, the parameterization is also connected with the form of space-time discretization that is used. In practice, each case has to be judged on its own merits: a model becomes 'over-parameterized' when clear signs of poor identifiability appear during model parameter estimation, although some methods of estimation or calibration are better at detecting this than others (see the example considered below).

3 The level and nature of noise on the data: In statistical parameter estimation, the uncertainty on the parameter estimates is a function of the noise/signal ratio on the data and the nature of this noise. If the noise level is high or the noise is not well behaved in some manner, then the parameter estimates may have high estimation error variance. This not only can produce effects that are very similar to those encountered when the model is poorly identifiable, but also it can affect the identifiability by attenuating the information content in the data.

4 Prior knowledge about model structure and parameter values: While it is clear that **exact** prior information on the model structure, as well as the value of certain parameters that characterize this structure, can enhance the model identifiability by reducing the number of parameters that have to be estimated, it is also clear that such a desirable situation never exists in the real world. For instance, the majority of assumed known parameters in such models have to be defined from *a priori* information, such as soil types and vegetation cover. But, unlike the situation in other areas of engineering, catchment hydrology modellers are not dealing with a well-defined 'man-made' system, so that the relevance of such parameters within the assumed model structure is often questionable. Consequently, the parameters normally have to be adjusted from their 'measured' values in order that the model is able to explain the data satisfactorily. This implies that they are not 'well known' when utilized in this modelling context and so, instead of being constrained to fixed values, they need to be considered as inherently uncertain.

Fortunately, with the increasing recognition of these inherent problems and the advent of computer-based, numerical stochastic techniques, such as MCS and sensitivity analysis, it has become increasingly common for hydrological modellers to assume that both the model structure and the associated parameters are uncertain and so defined by probability distributions, rather than point values (Beven 2009). Even with this welcome development, however, the problem of identifiability remains: as we see in the following illustrative example, if the model is over-parameterized, then the assumption of uncertainty in the parameters can tend to conceal rather than cure any underlying lack of identifiability.

An illustrative example of identifiability

In order to illustrate the concept of identifiability and its importance in real-time updating, let us look briefly at a real example, where the parameters of a daily rainfall-flow model are being updated each day, in real time, using a recursive estimation algorithm. The exact nature of these data, the associated model and the method of recursive estimation are not important at this time, since we are concerned only with the general consequences of over-parameterization and poor identifiability on real-time updating. However, in order to place the results in context, they relate to the analysis of daily effective rainfall-flow series from the Leaf River in Mississippi, USA, a segment of which is shown in Figure 9.3, where the effective rainfall is the rainfall adjusted nonlinearly to account for catchment storage effects. These series are part of a dataset that is used in the next section, which presents an illustrative example of forecasting and real-time state/parameter updating. Consequently, the identifiability results presented here are relevant to the selection of the rainfall-flow model used in this subsequent example.

Figure 9.4 compares the recursive estimates of all the assumed time-invariant parameters, as obtained over several years (a total of 1681 days) of data. The results for a reasonably identifiable, third-order model, are shown in the lefthand panel; and those for an over-parameterized and poorly identifiable seventh-order model, are plotted in the righthand panel. We see that the recursive estimates in the lefthand panel converge rapidly and, although they fluctuate to some degree, as one would expect with estimation from noisy data, this is far less than that encountered in the righthand panel. In order to examine this more closely, Figure 9.5 shows the estimation results for a typical parameter in each model, where the recursively updated estimate is displayed, together with its estimated standard error bounds. This illustrates how the estimated parameter in the higher order model not only fluctuates much more widely, but also has much wider uncertainty bounds.

In spite of the extremely volatile behaviour seen in the righthand panels of Figures 9.4 and 9.5, however, the model defined by the final estimates of the parameters explains the flow data to

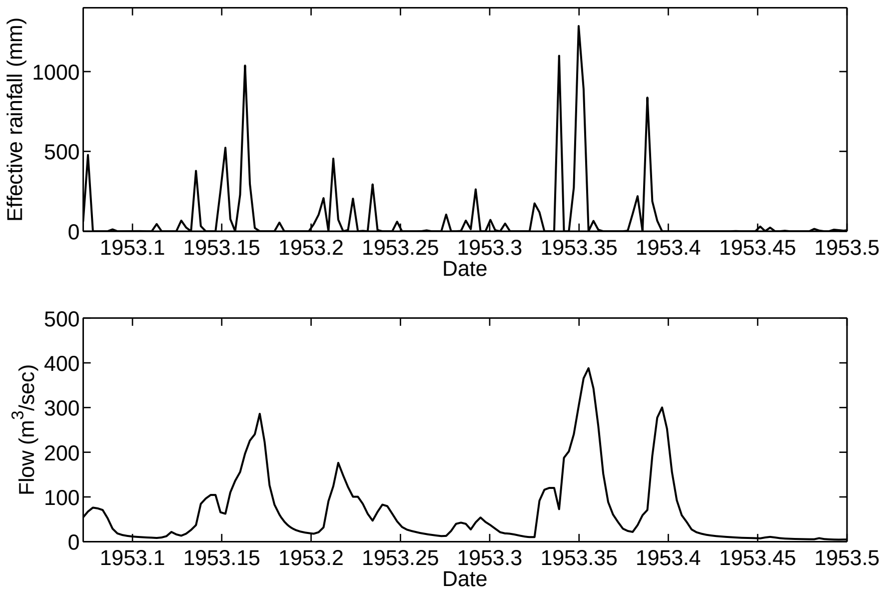

Fig. 9.3 A segment of the effective rainfall (top panel) and flow (lower panel) data from the Leaf River used in the illustrative example.

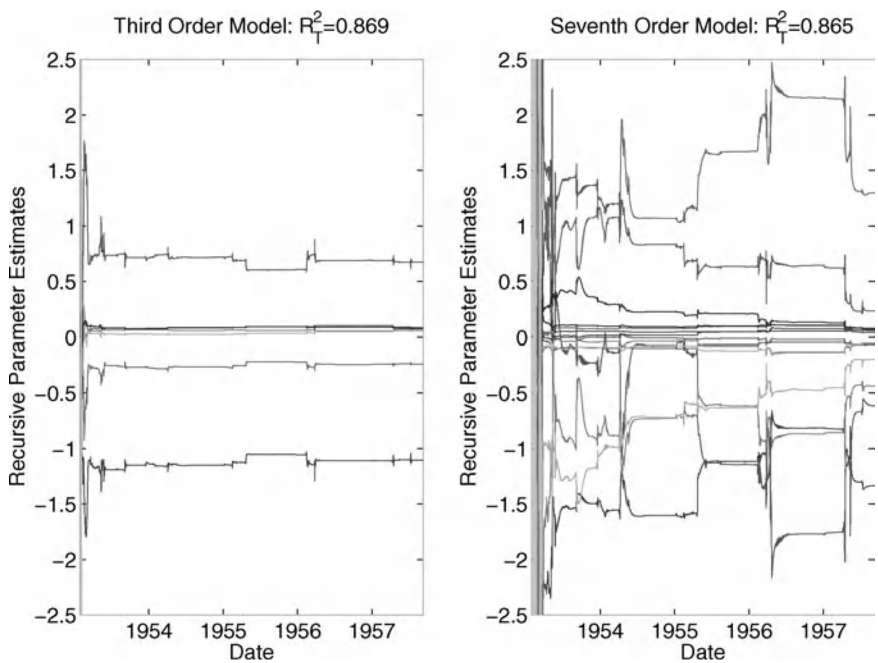

Fig. 9.4 Comparison of recursive estimates obtained for an identifiable third-order model (left panel) and a poorly identifiable seventh-order model (right panel). (See the colour version of this Figure in Colour Plate section.)

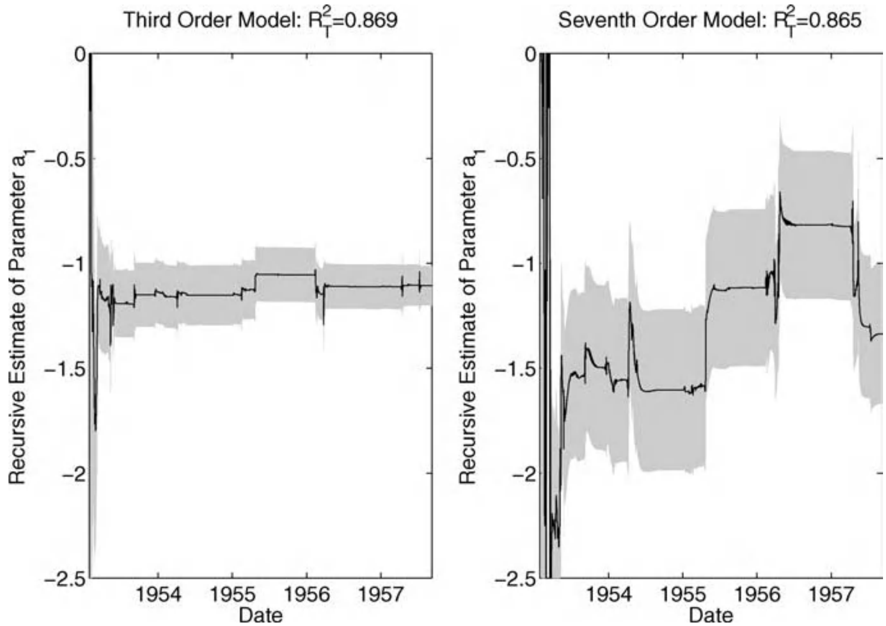

Fig. 9.5 Comparison of recursive estimates of the parameter a_1 obtained for an identifiable third-order model (left panel) and a poorly identifiable seventh-order model (right panel).

virtually the same extent as the well-identified model: the Nash–Sutcliffe efficency (equivalent to the coefficient of determination based on the simulated model response) is $R_T^2 = 0.864$ compared with $R_T^2 = 0.869$ for the third-order model. Moreover, the recursively updated seventh-order model quite often explains the data well at intermediate samples over the dataset, as we see in Figure 9.6. Here, the simulated response of the model obtained in the third month of 1955 ($R_T^2 = 0.869$) is compared with the responses of the seventh- and third-order models estimated at the end of the data, in the eighth month of 1957. All the models explain the data reasonably well but the two seventh-order models have substantially different estimated parameters.

When considering the recursive estimation results for the seventh-order model in Figure 9.4, it might be thought that the significant changes in the parameter estimates are redolent of nonstationarity in the model, with the parameters needing to change to reflect changing effective rainfall-flow dynamics. But this is clearly not the case, since the third-order model has an equivalent ability to explain the observed flow variation and its parameter estimates change very little. In other words, the estimation results in this over-parameterized situation are highly misleading. The importance of this phenomenon in terms of real-time updating is clear: while the recursively estimated parameters of the identifiable third-order model converge to reasonably well-defined values that change only a little over the whole of the data, the parameters of the over-parameterized, seventh-order model are highly volatile and never converge to stable values.

The fact that, for most of the observation interval, the estimated model explains the flow data as well as the third-order model, despite this volatility, shows that the estimates are inherently ambiguous and the model is poorly identifiable: a condition that has been termed 'equifinality' by von Bertalanffy (1968) and used in the hydrological literature by Beven and Freer (2001) and Beven (2006). Moreover, one of the eigenvalues of the seventh-order model is very close to the instability

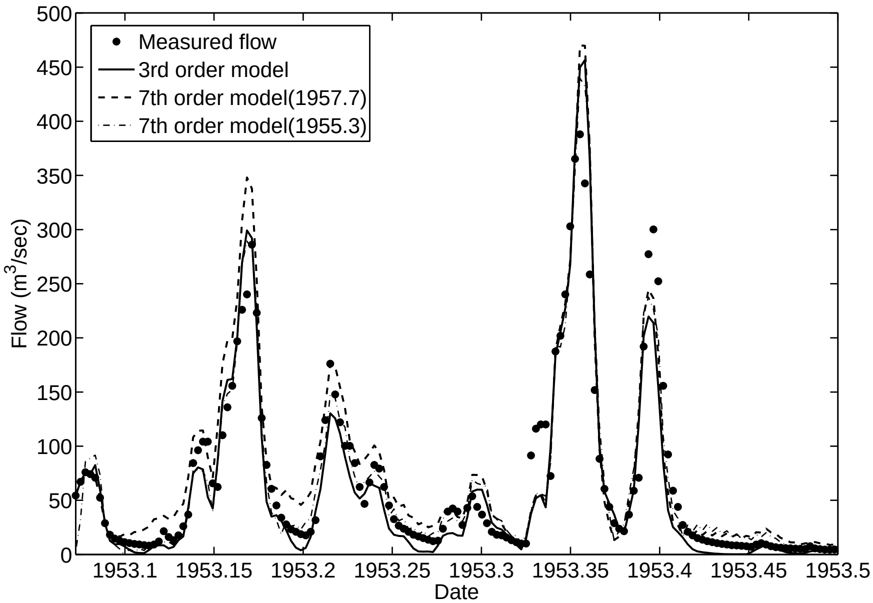

Fig. 9.6 Comparison of the simulated responses of the third-order estimated model and two seventh-order models estimated at different times.

boundary (indicating the estimation of a very long residence time); indeed, this sometimes becomes transiently unstable and has to be stabilized. Obviously, then, such a volatile and marginally stable model is not an appropriate vehicle for reliable flood forecasting.

As noted in factor 4 of the above list, it is possible to overcome the kind of over-parameterization induced identifiability problems shown in Figures 9.4 to 9.6, but only at the cost of imposing tight prior constraints on the parameter values, either deterministically, or stochastically. The latter stochastic, or 'Bayesian', approach is more sophisticated and defensible, since it recognizes overtly the inherent uncertainty in the parameter values. Figure 9.7 illustrates this process in the case of the seventh-order model considered above. Here, the left panel shows the same results as the right panel in Figure 9.4, but the right panel illustrates what happens if the model is estimated again on the same data but with the prior parameter estimate vector $\hat{\boldsymbol{\theta}}_0$ and associated covariance matrix $\mathbf{P}_{\boldsymbol{\theta},0}$ (see above) related to their finally estimated values on the left panel (i.e. as estimated after 1681 days): in particular the prior estimate $\hat{\boldsymbol{\theta}}_0$ is set exactly to $\hat{\boldsymbol{\theta}}_{1681}$, while the prior covariance matrix is set to $\mathbf{P}_{\boldsymbol{\theta},0} = 0.0001 \times \mathbf{P}_{1681}$, ensuring that there are very tight constraints on the subsequent recursive estimates (and so implying great confidence in the prior estimate $\hat{\boldsymbol{\theta}}_0$). As we see, this allows the estimates to fluctuate by a small amount in response to the information imparted by the latest effective rainfall-flow behaviour but they remain in the general location of their prior values. And at the end of the data, the explanation of the data has not changed much with $R_T^2 = 0.865$.[6]

The problem is, of course, that when confronted by the above results, the modeller **might** think that the model is well defined: it describes the data as well as the third-order model and the parameter estimates are stable and well defined. But this is

[6] The dependence of the recursive estimation results on the priors $\hat{\boldsymbol{\theta}}_0$ and $\mathbf{P}_{\theta,0}$ is well known and, normally, a diffuse prior ($\hat{\boldsymbol{\theta}}_0 = \mathbf{0}$; $\mathbf{P}_{\theta,0} = \text{diag} \times 10^6$) is assumed in order to avoid misleading estimation results.

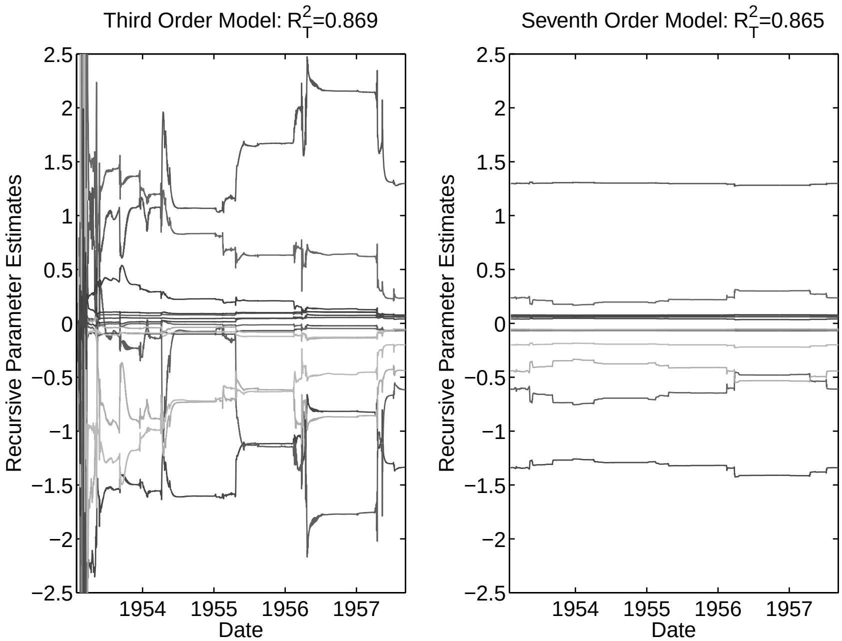

Fig. 9.7 The effect of imposing Bayesian prior constraints on the seventh-order model: no constraint with diffuse prior (left panel); tight constraint with Bayesian prior (right panel). (See the colour version of this Figure in Colour Plate section.)

illusory; it only occurs because very tight prior constraints have been imposed on the operation of the recursive estimation algorithm, implying much more confidence than justified in the *a priori* assumed values of the parameter estimates.

There are, of course, numerous high-order catchment simulation models used in hydrology, and so the Bayesian approach to estimating the parameters in such models is appealing and can be quite useful, provided it is applied with care, recognizing that it is very dependent on the validity of the assumed prior knowledge. Unfortunately, this does not always seem to be the case.

Data Assimilation and Adaptive Forecasting: An Illustrative Tutorial Example

As pointed out earlier, this example is concerned with daily rainfall-flow data from the Leaf River catchment, a humid watershed with an area of 1944 km^2 located north of Collins, Mississippi, USA. These data have been selected because they have been used as the basis for recent research on the application of some of the newest methods of data assimilation mentioned previously: the EnKF and the PF (Moradkhani *et al.* 2005a, 2005b; Smith *et al.* 2006). Consequently, the results presented below can be viewed in the context of these previous studies and the methods that they describe.[7]

For simplicity of presentation and in order that more detailed aspects of the model estimation and forecasting system design process can be emphasized, the example concerns only a single model relationship between rainfall and flow at a single site on the Leaf River. When this methodology is

[7] Note that, in these papers, the term 'residence time' is used incorrectly when referring to the **inverse** of the residence time, with inverse time units (d^{-1}); the residence time referred to in the present section is defined in the standard manner, with daily time units.

applied to a whole river basin, however, a simple model such as this would form just one element in a quasi-distributed model of the total catchment network. A typical example is the Lancaster forecasting system for part of the River Severn (Romanowicz *et al.* 2006; Young *et al.* 2006), which contains additional rainfall-flow models of this type, as well as linear and nonlinear flow routing models. The same methodological approach to that described below is used for each of these submodels, prior to their assembly in the complete catchment model and forecasting system. This approach is currently being used in the design of a forecasting system for the River Eden catchment in the UK that is being incorporated into the Delft-FEWS scheme as part of a project funded by the UK's Environment Agency in connection with the development of its National Flood Forecasting System (NFFS: see also Beven *et al.* 2008; Young *et al.* 2009).

Despite its simplicity, the example presents a difficult flow forecasting exercise. Indeed, it has been selected because these difficulties help to illustrate various important aspects of forecasting system design. For instance, the example is concerned with daily data and yet there is no advective delay δ between the occurrence of rainfall and its effect on flow. Indeed, there is a significant **instantaneous** effect (i.e. a flow effect resulting from rainfall falling within the same day). Clearly, therefore, one-day-ahead flow forecasting, without an accompanying one-day-ahead forecast of the rainfall, presents quite a challenge. In addition, the imposition of constraints on the model structure so that it has real eigenvalues and is similar to other previous models used with these data, means that the model may not be as good in forecasting terms as it could have been with the statistically identified, unconstrained structure.

The Data-Based Mechanistic (DBM) model

In the previous studies mentioned above, the conceptual fourth-order HYMOD model, as shown diagrammatically in Figure 9.8 with one slow-flow plus three quick-flow tanks, is used to evaluate the various data assimilation algorithms. In order to aid comparison, the Data-Based Mechanistic (DBM) model of the Leaf River used in the present example is identified in a form that resembles the HYMOD model. In particular, the model is constrained to be third order with real eigenvalues, even though statistical identification and estimation analysis suggests that a second- or third-order model with the naturally estimated complex

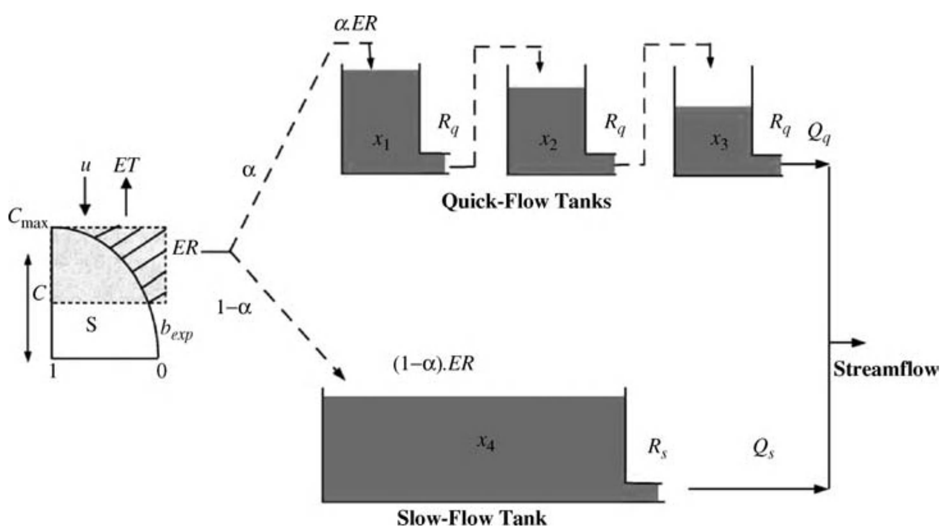

Fig. 9.8 The HYMOD model of the Leaf River.

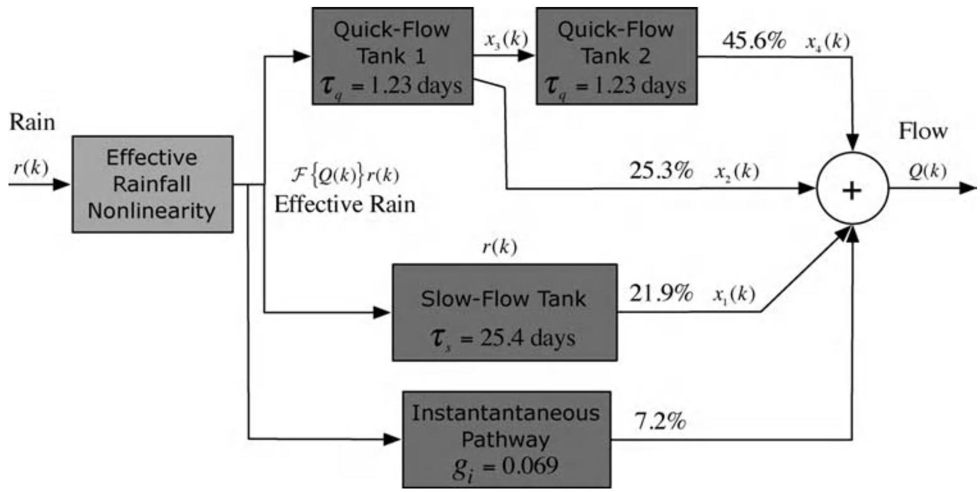

Fig. 9.9 The Data-Based Mechanistic (DBM) model of the Leaf River: here, τ_q and τ_s are the quick and slow residence times, while the percentage figures denote the percentage of flow passing down the indicated pathways. (See the colour version of this Figure in Colour Plate section.)

eigenvalues may be superior in statistical terms. A fourth-order model with three quick-flow tanks was rejected because of clear identifiability problems.

The identified third-order, constrained DBM model was estimated from an estimation dataset of 366 days over 1952–1953 and is shown diagrammatically in Figure 9.9, where the output flow is now denoted by Q_k, rather than the more general output variable y_k used in previous sections, in order to aid comparison with the HYMOD diagram in Figure 9.8: both have an input 'effective rainfall' nonlinearity, although these are not of the same form; and both have a 'parallel pathway' flow structure consisting of 'quick' and 'slow' flow tanks, although HYMOD has the additional quick-flow tank, mentioned above, which is effectively replaced in the DBM model by the 'instantaneous' pathway, represented by a simple gain coefficient with no dynamics.

The main difference between the models is that whereas in HYMOD the *a priori* conceptualized structure is fitted to the data in a hypothetico-deductive manner, the DBM model structure is identified statistically from the data in an inductive manner, without prior assumptions other than that the data can be described by a nonlinear differential equation or, as in this case, an equivalent discrete-time difference equation: see, for example, the discussion in Young (2002). It is necessary to stress again that the DBM model here is constrained to some extent in order to match the structure of the HYMOD model and this would not normally be the case in DBM modelling (see discussion under 'Comments' below).

The DBM model is estimated initially in a nonlinear Transfer Function (TF) form since it is possible to do this by exploiting easily available, general TF estimation tools, such as those in the CAPTAIN Toolbox mentioned previously. This contrasts with direct estimation of a specified model in state space model form, where a customized algorithm is required. Moreover, it can be shown that there are statistical advantages in estimating the minimally parameterized transfer function model and it can be transformed easily into **any** selected state space model form that has relevance to the problem at hand: for example, one that has a useful physical significance. The DBM model in Figure 9.9 is obtained in this manner and it relates to the following state space form, which was obtained from the estimated TF model with the help of the *residue* routine in Matlab:

$$\begin{bmatrix} Q_{1,k} \\ Q_{2,k} \\ Q_{3,k} \\ Q_{4,k} \end{bmatrix} = \begin{bmatrix} f_{11} & 0 & 0 & 0 \\ 0 & f_{22} & 0 & 0 \\ 0 & 0 & f_{32} & f_{33} \\ 0 & 0 & 1 & 0 \end{bmatrix} \begin{bmatrix} Q_{1,k-1} \\ Q_{2,k-1} \\ Q_{3,k-1} \\ Q_{4,k-1} \end{bmatrix} + \begin{bmatrix} g_1 \\ g_2 \\ g_3 \\ 0 \end{bmatrix} u_k + \begin{bmatrix} \eta_{1,k} \\ \eta_{2,k} \\ \eta_{3,k} \\ \eta_{4,k} \end{bmatrix} \quad (9.7a)$$

with the associated observation equation:

$$Q_k = [1 \quad 1 \quad 0 \quad 1] \begin{bmatrix} Q_{1,k} \\ Q_{2,k} \\ Q_{3,k} \\ Q_{4,k} \end{bmatrix} + g_i u_k + e_k \quad (9.7b)$$

Here, the state variables $Q_{i,k}$, $i = 1, 2, 4$ are, respectively, the 'slow flow' $Q_{1,k}$, normally associated with the groundwater processes; and the two 'quick' flows, $Q_{2,k}$ and $Q_{4,k}$, normally associated with the surface and near-surface processes. The fourth state, $Q_{3,k}$, is an intermediate state arising from the transfer function decomposition shown in Figure 9.9. These are all 'unobserved state variables' that have to be estimated by the KF algorithm when the model is used in forecasting. As we see in the observation equation (Equation 9.7b), the flow measurement is the sum of the first, second and fourth of these state variables, plus an instantaneous term dependent on rainfall occurring during the same day.

The effective rainfall input u_k is defined as $uk = F(Q_k)r_k$ where $F(Q_k)$ is the estimated State-Dependent Parameter (SDP) nonlinearity (Young 2001b, 2003; Young *et al.* 2001), which operates on the measured rainfall r_k to yield the effective rainfall u_k. Note that the sampling index associated with this term in the above state equations is k, indicating that there is no advective delay between the occurrence of rainfall and its effect on flow, as pointed out previously. As we shall see later, this needs to be taken into account when the model is utilized for flow forecasting: in particular, the estimated model (Equations 9.7a and 9.7b) needs to be modified so that a false one-day advective delay is introduced (u_k changed to u_{k-1}) and the instantaneous effect is set to zero ($g_i = 0$). Note also that this SDP nonlinearity serves a similar role to the conceptual effective rainfall nonlinearity in the HYMOD model, except that the SDP nonlinearity is a function of the measured flow Q_k, here acting as a surrogate measure of the catchment storage (Young 2002). In the HYMOD model, the effective rainfall nonlinearity is of the PDM type (Moore 1985) and it makes use of potential evapotranspiration data, which was not utilized at all in the present DBM model.

The model (Equations 9.7a and 9.7b) is estimated in a simple, two-stage statistical procedure, as discussed in previous publications (see, e.g., Young 2003; Young *et al.* 2007). First, a nonparametric (graphical) estimate of the input effective rainfall nonlinearity is obtained using the *sdp* estimation routine in the CAPTAIN Toolbox for Matlab (see footnote 4). This estimate is shown as the full black line in Figure 9.10, together with its associated 95% confidence region (grey). At the

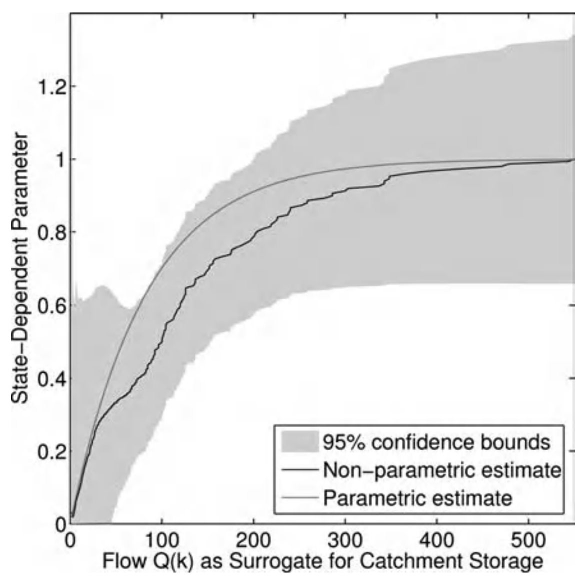

Fig. 9.10 Nonparametric (black line) and parametric (red line) estimates of the estimated State-Dependent Parameter (SDP) effective rainfall nonlinearity. (See the colour version of this Figure in Colour Plate section.)

second stage of the estimation, a suitable parameterization of this nonparametric estimate is then selected and the whole model is re-estimated using this parameterization.

In the present case, it is found that an exponential function,

$$F(Q_k) = \lambda\left(1 - e^{-\gamma Q_k}\right) \tag{9.7c}$$

provides a reasonable parameterization. Here, λ is a scaling factor selected to control the magnitude of the effective rainfall and, if required, ensure conservation of mass within the system. Note that Equation 9.7c is not an approximation of the estimated nonparametric function in Figure 9.10: the latter simply aids in the identification of a suitable parametric function, which is then estimated separately during the final estimation of the whole parameterized model.

The final estimation of the complete model (Equations 9.7a–9.7c) can be carried out in various ways but here the parameter vector

$$\boldsymbol{\theta} = [f_{11}\ f_{22}\ f_{32}\ f_{33}\ g_1\ g_2\ g_3\ g_i\ \gamma] \tag{9.8}$$

is estimated as follows by simple nonlinear least squares using the *lsqnonlin* routine in Matlab:

$$\hat{\boldsymbol{\theta}} = \arg\min_{\boldsymbol{\theta}} J(\boldsymbol{\theta}) \quad J(\boldsymbol{\theta}) = \sum_{k=1}^{N}\left[Q_k - \hat{Q}_k(\boldsymbol{\theta})\right]^2 \tag{9.9}$$

where $\hat{Q}_k(\boldsymbol{\theta})$ is the deterministic output of the model.

The details of this optimization procedure are as follows for an optimization step index *t:*
while the change in the estimate $\hat{\boldsymbol{\theta}}(t) - \hat{\boldsymbol{\theta}}(t-1) \geq \varepsilon$;

1 Update the parameter γ and use this to define the effective rainfall $u_k = F(Q_k)r_k$.

2 Update the remaining parameters using the *riv* algorithm in the CAPTAIN Toolbox, based on this effective rainfall and measured flow.

where ε is a user-specified convergence condition.

This optimization procedure yields a nonlinear TF model and the parameters of the state space model are then inferred from the TF model parameters by transformation, using the *residue* routine in Matlab. The estimates obtained in this manner, when applied to the estimation dataset over 1952–1953 are as follows:

$$\begin{aligned}
&\hat{f}_{11} = 0.9613(0.008);\ \hat{f}_{22} = 0.44449(0.010);\\
&\hat{f}_{33} = 0.8898(0.020) \quad \hat{f}_{34} = -0.1979(0.009);\\
&\hat{g}_1 = 0.00810(0.006);\ \hat{g}_2 = 0.13467(0.007)\\
&\hat{g}_3 = 0.13472(0.003);\ \hat{g}_i = 0.0691(0.004);\\
&\hat{\gamma} = 0.0121(0.0005)
\end{aligned} \tag{9.10}$$

where the figures in parentheses are the standard errors associated with the parameter estimates. The SDP nonlinearity defined by Equation 9.7c with the above estimated $\hat{\gamma}$ is shown as the red line in Figure 9.10, where we see that it is consistent with the initial nonparametric estimate. The model (Equations 9.7a and 9.7b) with the above parameter estimates constitutes the prior, 'nominal' DBM model of the Leaf River data.

The stochastic inputs to the model also have to be specified. In this connection, the $\eta_{i,k}$, $i = 1, 2, 3, 4$ in Equation 9.7a are assumed to be real stochastic inputs to the system, in this case unmeasured in-flow and out-flow effects, that lead to changes in the state variables; while e_k are the observational errors associated with the output flow measurement in Equation 9.7b. For the purposes of the present example, the observational errors will be assumed to have zero mean value and a changing variance in order to allow for the heteroscedastic effects that are known to affect flow data. In particular, SDP estimation shows that the variance can be represented by the following simple function of Q_k^2:

$$\sigma_k^2 = \beta\, Q_k^2 \tag{9.11}$$

where $\beta = 0.01$ provides a reasonable scaling with the associated standard deviation ranging from 0.2 at low flows to 40 m^3/s at the highest flow over the estimation year of 1953. This compares with the range of 0.01 to 80 m^3/s cited by Vrugt *et al.* (2005) for the Leaf River data (see also below, under 'State updating by the Kalman Filter'). The stochastic inputs, as defined by the vector $\boldsymbol{\eta}_k$, are assumed to be zero mean variables with the following

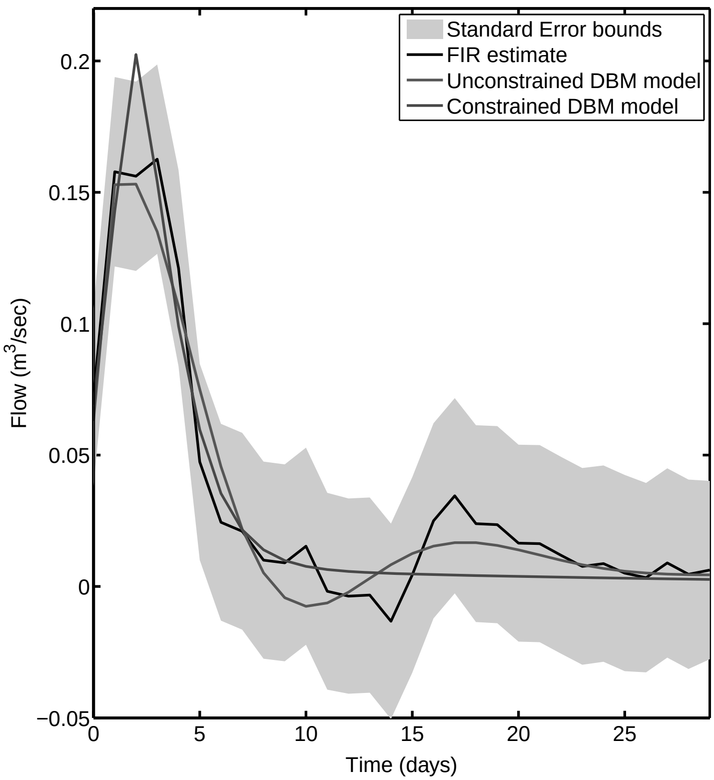

Fig. 9.11 Comparison of various unit hydrographs (impulse responses). (See the colour version of this figure in Colour Plate section.)

statistical properties:

$$\boldsymbol{\eta}_k = \left[\eta_{1,k}\ \eta_{2,k}\ \eta_{3,k}\ \eta_{4,k}\right]^T; \quad \boldsymbol{E}\left\{\boldsymbol{\eta}_j\,\boldsymbol{\eta}_k^T\right\} = \mathbf{Q}\delta_{kj} \tag{9.12}$$

where δ_{kj} is the Kronecker delta function $\delta_{kj} = 1.0$, $k = j$; $\delta_{kj} = 0$, $k \neq j$. Here, the covariance matrix $\mathbf{Q}$ is assumed to be constant over time and diagonal in order to simplify the final optimization of the forecasting system. Note that, as pointed out earlier, $\mathbf{Q}$ could have been made time variable and the heteroscedastic effects could have been linked with this. The quantification of the hyperparameters is discussed below, when we consider the KF forecasting system design.

The instantaneous effect in the observation equation arises from the data-based identification of rainfall-induced flow changes occurring **within** the daily sampling interval. This is illustrated in Figure 9.11, where a nonparametric estimate of the unit hydrograph[8] (full black line with 99% confidence bounds shown grey) clearly reveals the instantaneous jump at $k = 0$, which is confirmed by the unit hydrographs associated with the constrained model (Equations 9.7a and 9.7b) and the unconstrained DBM model. It is well known that forecasting daily flows is never easy, since the information in the daily rainfall-flow data is rather limited. In the present case, this situation is exacerbated further by this instantaneous effect, which means that there is no advective delay at all between the occurrence of rainfall and its first effect on the flow.

In order to utilize the model for forecasting in the above estimated form, it would be necessary to

[8] Here defined as the response to a unit input of effective rainfall applied at $k = 0$ and obtained by Finite Impulse Response (FIR) estimation.

have access to a one-day-ahead rainfall forecast, which is not available in this case. As a result, it is necessary to change the sampling index on the effective rainfall in Equation 9.7a to $k-1$ and ignore the second term in Equation 9.7b. In this manner, the KF is producing a true one-day-ahead forecast based only on the rainfall-flow measurements available at the time the forecast is computed. On the other hand, the parameter updating is based on the nominal estimated model form, without these changes, so ensuring that the model is always able to explain the data as well as possible.

Parameter updating by recursive RIV estimation

The model (Equations 9.7a and 9.7b) constitutes the 'nominal' model introduced in an earlier section, with the state space matrices $\mathbf{F}_0$ and $\mathbf{G}_0$ defined as:

$$\mathbf{F}_0 = \begin{bmatrix} \hat{f}_{11} & 0 & 0 & 0 \\ 0 & \hat{f}_{22} & 0 & 0 \\ 0 & 0 & \hat{f}_{33} & \hat{f}_{34} \\ 0 & 0 & 1 & 0 \end{bmatrix} \quad \text{and} \quad \mathbf{G}_0 \begin{bmatrix} \hat{g}_1 \\ \hat{g}_2 \\ \hat{g}_3 \\ 0 \end{bmatrix} \qquad (9.13)$$

where the parameter estimates are those given in Equation 9.10. In order to allow for parameter updating, we need to introduce a capacity for updating the parameters of this model as additional daily rainfall-flow data arrive. This can be introduced in various ways, as outlined earlier (see 'Parameter updating' above). Here, however, the recursive form of the RIV algorithm used at the nominal model parameter estimation stage is employed to update the parameters of the DBM transfer function model; the parameters in the state space model are then obtained from these by transformation, on a continuing basis. The recursive RIV estimation could include the continual updating of the parameter γ in the SDP effective rainfall nonlinearity (Equation 9.7c) but this did not affect the forecasting ability very much and was maintained at its nominal value. An alternative recursive estimation approach, in this case, would be to use the Lin and Beck RPE algorithm (see 'Parameter updating' above), configured specially for the state space model (Equations 9.7a and 9.7b).

Finally, note that the separate parameter estimation approach used here can be contrasted with that used by Moradkhani *et al.* (2005b) and Smith *et al.* (2006) in their research on the Leaf River, where the state of the EnKF is extended to include model parameters and both are estimated concurrently and interactively (see previous discussion on this under 'Parameter updating' above).

State updating by the Kalman Filter (KF)

Given the model (Equations 9.7a and 9.7b) and the associated stochastic hyper-parameter definitions in Equations 9.11 and 9.12, the KF described by Equations 9.4a to 9.4f provides an obvious starting point for the design of a real-time forecasting engine. In order to utilize this, it is necessary to quantify the various hyper-parameters that control the KF forecasting performance; namely, the observation noise variance σ_k^2 in Equation 9.11 and the diagonal elements of the stochastic input covariance matrix $\mathbf{Q}$ in Equation 9.12. And if parameter updating is required, then the parameter δ_θ that defines the covariance matrix $\mathbf{Q}_\theta$ for recursive parameter estimation, as discussed above, is also required.

Although all of these hyper-parameters could be optimized, this was not attempted in this case so that the ease of manual selection could be demonstrated. The diagonal elements of $\mathbf{Q}$ were selected on the basis of the empirically estimated diagonal elements of the state variable covariance matrix, computed over the same data as those used for the nominal model estimation. These were then normalized around the second state variance and $\mathbf{Q}$ defined as follows:

$$\mathbf{Q} = \text{diag}\big(\delta_q[0.634 \quad 1.0 \quad 3.245 \quad 0]\big) \qquad (9.14)$$

where δ_q is now the only hyper-parameter to be determined.

At this point it is necessary to consider the relative levels of uncertainty in the system. If we assume that the random errors in the flow measurements are small relative to the uncertainty associated with the stochastic inputs to the state equations, then δ_q should be very much greater than the observation noise hyper-parameter $\beta = 0.01$ in Equation 9.11, which defines an appropriate level of uncertainty on the flow observations and the associated one-day-ahead forecasts. As δ_q is increased, so the forecasting performance improves but there is little further improvement after $\delta_q = 10{,}000$, which is, therefore, the selected value. Finally, it was found that a very small value of $\delta_\theta = 10^{-14}$ for the parameter tracking hyper-parameter yielded the most acceptable tracking results (see 'Comments' below).

Typical adaptive forecasting results

Typical adaptive forecasting results are presented in Figures 9.12 to 9.15. These are obtained with recursive parameter updating applied only to the

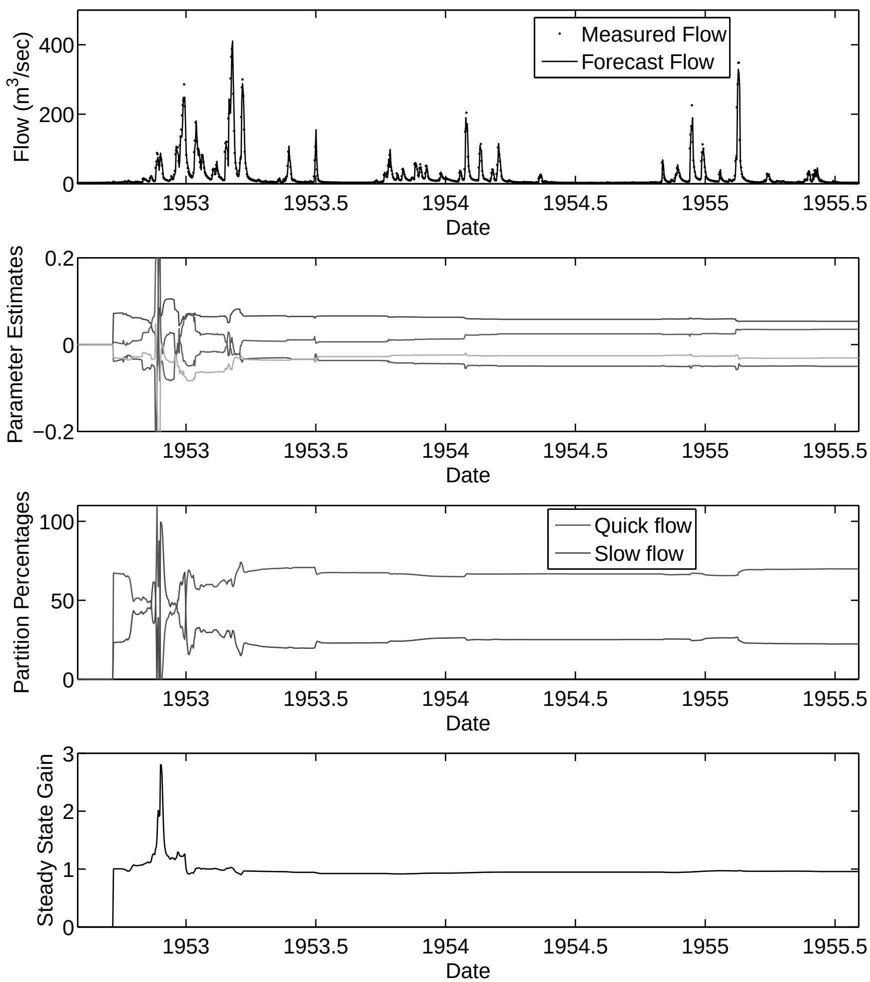

Fig. 9.12 Leaf River example: three years of real-time updating following initiation after 50 days: measured and forecast flow (upper panel); recursive estimates of model parameters (upper middle panel) and partition percentages (lower middle panel); and overall steady state gain (lower panel). (See the colour version of this Figure in Colour Plate section.)

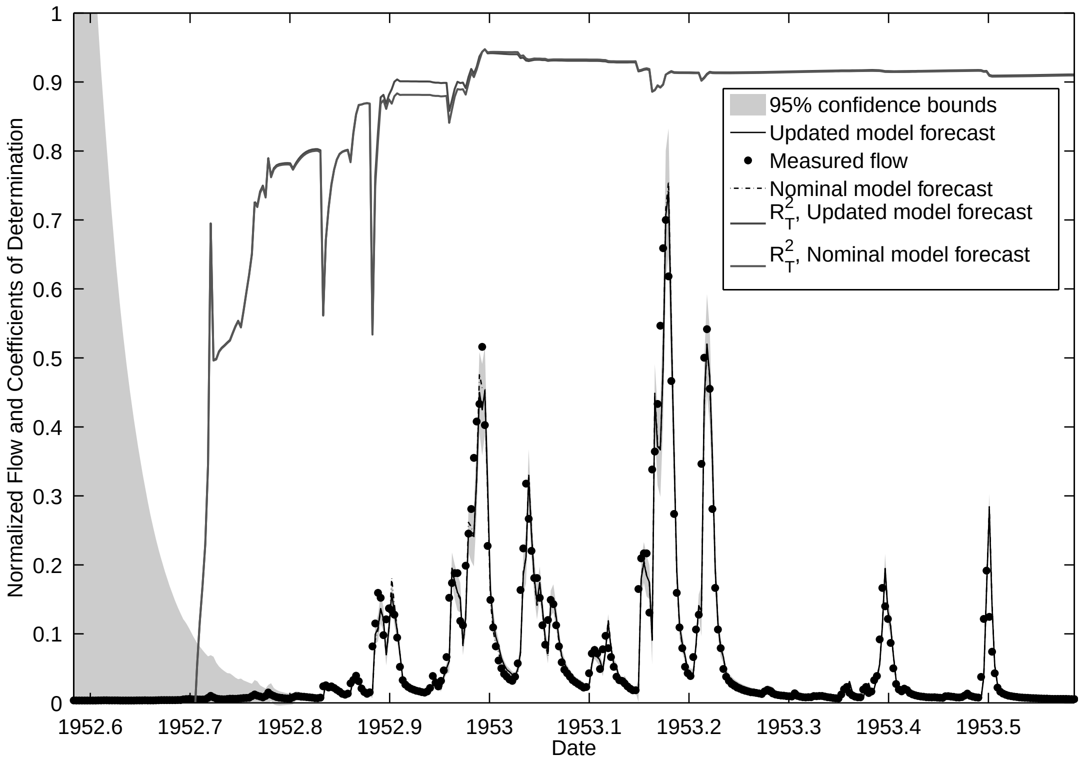

Fig. 9.13 Leaf River example: more detailed view of the real-time updated forecasting over the first year showing estimated 95% confidence bounds and running mean R_T^2 values for updated and fixed model forecasts, based on the innovation errors. (See the colour version of this Figure in Colour Plate section.)

numerator coefficients in the transfer function, with the denominator coefficients maintained at their nominal estimated values. To ensure this, the initial covariance matrix for the recursive RIV parameter estimation $\mathbf{P}_{\theta,0}$, is set to reflect some considerable uncertainty in the numerator parameters but no uncertainty in the denominator parameters (i.e. the relevant elements of $\mathbf{P}_{\theta,0}$ are set to zero). In effect, this is informing the algorithm that we are confident in the constrained eigenvalues of the nominal model and the associated residence times of the flow pathways, but we are not sure that the steady-state gains and the consequent partition percentages of flow in these pathways will not change over time.

Figure 9.12 shows three years of real-time updating following initiation after 50 days. Here, since the initial covariance matrix for the recursive RIV parameter estimation is set to reflect some considerable uncertainty in the parameters, the estimates are rather volatile when the first large rainfall and flow events occur. In particular, the recursive estimates of the four updated model parameters (i.e. the coefficients of the numerator polynomial in the transfer function), as plotted in the upper middle panel, vary quite a lot while the RIV estimation algorithm is 'learning' the model parameters from the rainfall-flow data. However, after this is completed early in 1953, they then settle down to become fairly stable when sufficient information has been processed to engender confidence in the estimates. Note that the associated changes in the parameters of the state space model (Equation 9.7a) can be inferred

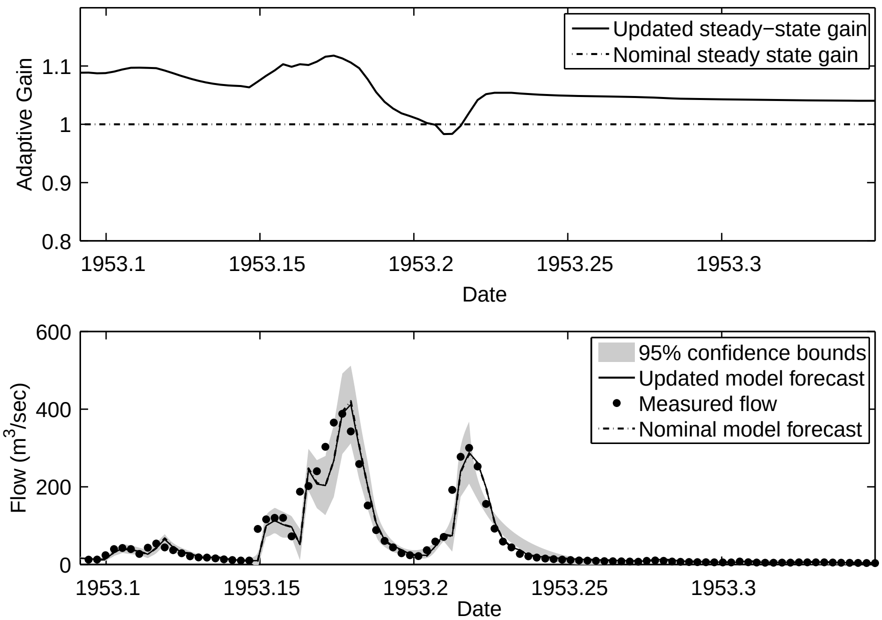

Fig. 9.14 Leaf River example: the lower panel is a short part of Figure 9.13 showing more clearly the forecasting performance. The changes in the continually updated estimate of the model steady-state gain (top panel) make little difference to the model forecast (full-line) in comparison with that of the constant parameter nominal model (dash-dot line).

straightforwardly from the changes in these estimated transfer function parameters, and the same estimation behaviour is reflected in the associated changes in the partition percentages, shown in the lower middle panel. Here, for clarity, the quick-flow percentage is obtained by aggregating the percentages of the two estimated quick-flow pathways. The overall steady-state gain, plotted in the lower panel, also shows little change after the learning period is complete.

Figure 9.13 presents a more detailed view of the real-time updated forecasting over the first year, showing estimated 95% confidence bounds and running mean values of the coefficient of determination R_T^2 (Nash–Sutcliffe efficiency) for the updated (blue line) and fixed parameter, nominal model (red line) forecasts. The improvement is very small because the nominal model clearly provides a good representation of the rainfall-flow dynamics over the whole of the time period and there is little need for significant adaption: in effect, both fixed and adaptive models are performing in a quite similar manner over most of the time period.

Figure 9.14 presents a still more detailed view of the adaptive forecasting performance. The lower panel is a short segment of Figure 9.13 showing more clearly the effect that real-time model parameter updating has on the forecasting performance. The estimated 95% confidence interval is consistent with the forecasts and captures the heteroscedastic behaviour of the forecasting errors, except during the upward part of the hydrograph, where the flow measurement is sometimes marginally outside this interval. This is a direct consequence of the forecasting problems, mentioned previously, caused by the absence of any advective time delay between the rainfall and flow in this example: the forecasting system has no prior warning of the rainfall and it is impossible

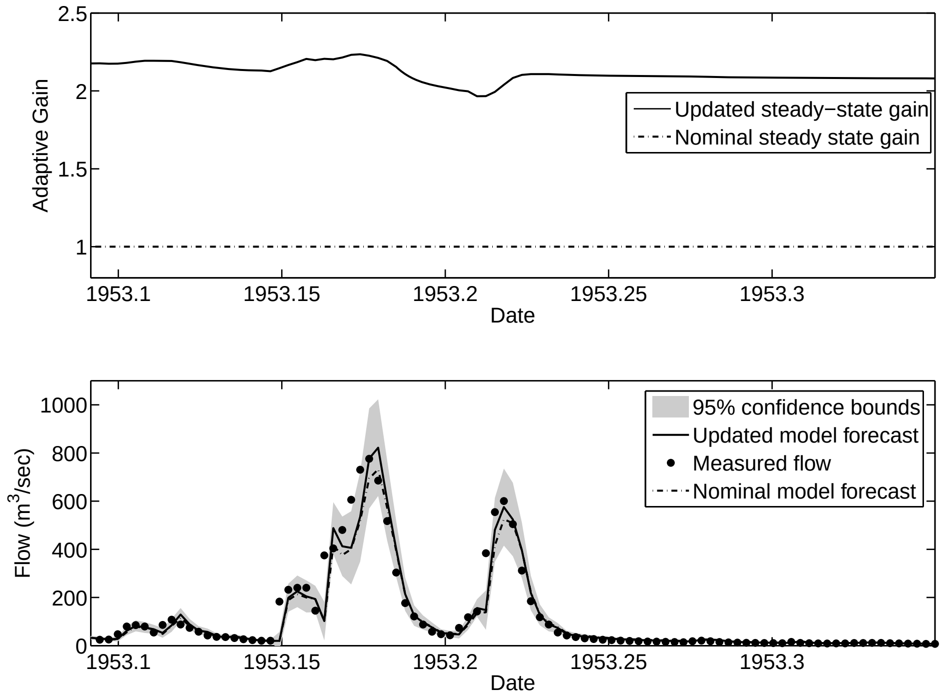

Fig. 9.15 Leaf River example: repeat of Figure 9.14 but with the measured flow doubled in magnitude so that the nominal model now has a substantial steady state gain error. The continually updated gain estimate correctly identifies the effective doubling of the gain and the adaptive forecast is better than the nominal model forecast, although the difference is still not substantial.

for it to forecast the associated change in flow any better than this unless a one-day-ahead rainfall forecast is also available.

Finally, Figure 9.15 shows what happens if the rainfall-flow dynamics are not captured well by the nominal model. In particular, the flow measurements are doubled in magnitude, so that the nominal model gain is now considerably in error. It is clear that the recursively estimated steady-state gain has responded to the change in the flow data and is about double the estimate in Figure 9.14. Perhaps more surprisingly, at first sight, the resulting difference in the adaptive and nominal model forecasts is still not all that significant: the adaptive forecast is better but the differences are only visible around the peaks of the hydrograph. This demonstrates the robustness of the KF, particularly in this situation when the observational errors are assumed to be low (with a small associated hyper-parameter $\beta = 0.01$ relative to the stochastic input hyper-parameter $\delta_q = 10{,}000$), so that the output estimate is brought back closely to the region of the flow measurement at each corrective update.

Comments

1 Given the nature of the unit hydrograph plots of Figure 9.11, with the absence of any advective time delay and a significant instantaneous effect of rainfall on flow, it is not surprising that the immediate forecasts on the upward part of the hydrograph, following a rainfall event, are poorer than those on the recession part of the hydrograph.

One possibility is to estimate the model directly in the form used for forecasting, with the false time delay included in the model and no instantaneous term: this naturally reduces the explanatory power of the model, in relation to the model (Equations 9.7a and 9.7b), and makes it less realistic in physical terms, but it can sometimes improve forecasting performance (Lees *et al.* 1994; Young 2002). In this case, however, the overall forecasting performance is not improved.

2 One limitation of the KF-based forecasting scheme used in this example is that the forecast does not take into account the uncertainty of the parameter estimates, so that the uncertainty bounds are theoretically a little too narrow. Allowance for such uncertainty could be introduced but the additional uncertainty is very small in the case of the statistically efficient DBM model used here, so it was not considered necessary.

3 Theoretically, the innovations sequence produced by the KF should be a zero mean, serially uncorrelated white noise sequence. In the present example, there is a just significant autocorrelation of 0.19 at a lag of 1 day, which could be corrected by adding a stochastic state variable to account for this, based on an **AutoRegressive** AR(1) model. However, the correlation is quite small and this modification makes little difference to the forecasting performance. So, as in point 2, above, there seems little reason to complicate the algorithm in this case.

4 The computational cost of the adaptive forecasting implementation described here is very small: each daily state/parameter update takes only a few microseconds on a standard desktop computer and even this could be improved considerably by efficient programming. In contrast, the numerically intensive alternatives, such as the EnKF and PF, are inherently more computationally expensive, requiring a specified number of Monte Carlo realizations within each update ensemble: for example, Moradkhani *et al.* (2005b) cite an ensemble of 40 realizations, each requiring solution of the model equations. It is not clear, therefore, what is gained by this additional computational load in the present example because the EnKF only provides an approximate, numerical solution of the linear KF equations, which, as we have seen, are clearly an alternative in this case because the only nonlinearity in the HYMOD model is the effective rainfall nonlinearity at the input. Of course, in other situations, where the selected model is large and there are high levels of **internal** nonlinearity, then computationally intensive algorithms, such as the EnKF and the similar numerically intensive methods, as discussed earlier (see Large and highly nonlinear stochastic systems), provide a sensible approach.

5 Finally, the continuing verification of the forecasting performance is important in practice. This is aided by the adaptive nature of the recursive algorithms employed here, where the user is continually informed of the updated parameters and states and is able to assess performance. This can be enhanced by the computation and presentation of related performance measures: for example, plots of statistics such as Root-Mean-Square Error (RMSE), Mean Absolute Percentage Error (MAPE) and other measures of skill against multiple forecast lead-times.

Conclusions

This largely tutorial chapter has considered the state of the art in the real-time updating of states and parameters in flood forecasting models. Given the enormous number of publications on this topic, there has been no attempt to review all of the available techniques that are currently available. Rather the chapter has concentrated on those techniques that have come into prominence during the last few years, and has addressed some important topics raised by these developments, including: the problem of model identifiability and its effect on time-variable parameter estimation; the relative merits of joint and separate state-parameter estimation in real-time updating; and the choice between analytic or computationally intensive methods of recursive state estimation and forecasting.

The chapter concentrates on lumped parameter models that can be used in the development of quasi-distributed flood forecasting and warning systems but it does not address directly the question of real-time updating in fully distributed models, such as detailed hydrodynamic representation of a catchment, the grid-to-grid models of Moore and his co-workers (see, e.g., Moore *et al.* 2006; Cole and Moore 2008) and other types of distributed hydrological models. Clearly, the problems of identifiability severely restrict attempts at updating the parameters of such large, over-parameterized models unless some model-specific procedures are invoked to constrain the ill-posedness of the problem. In principle, however, the concepts and methods for state updating outlined here are applicable to such large models and can be applied to them provided the associated state variables are observable from the available rainfall-flow or level data. Unfortunately, the complexity of large hydrodynamic models is such that observability is difficult to guarantee, so that a truly systematic approach, such as this, is rarely possible and ad hoc, partial solutions are normally required. These are difficult to generalize since they depend so much on the specific nature of the model. A recent example is the use of the EnKF and EnSRF with the distributed hydrological model **TopNet** by Clark *et al.* (2008), where the authors report many problems and the results are not particularly good. They conclude that: 'New methods are needed to produce ensemble simulations that both reflect total model error and adequately simulate the spatial variability of hydrological states and fluxes.'

One possible approach to real-time updating in the case of large models is the idea of emulating a large dynamic simulation model by a small data-based mechanistic (DBM) model or some other form of low-dimensional emulation model. As Young and Ratto (2008) have suggested, such emulation models can also help to provide a unified approach to stochastic, dynamic modelling that combines the hypothetico-deductive virtues of good scientific intuition, as reflected in the large hydrological or hydrodynamic simulation model, with the pragmatism of inductive DBM modelling, where more objective inference from data is the primary driving force.

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10 Coupling Meteorological and Hydrological Models for Real-Time Flood Forecasting

GEOFF AUSTIN, BARNEY AUSTIN, LUKE SUTHERLAND-STACEY AND PAUL SHUCKSMITH

Overview

Radar and satellite data have been used successfully for years in rainfall and flood forecasting. In fact, nowcasting (i.e. short-range forecasts based largely on the extrapolation of current information) forms the backbone of many real-time flood warning systems, particularly for small urban catchments where the time between rainfall and serious flooding can be short. The use of meteorological Numerical Weather Prediction (NWP) is less common, but recently these models have undergone major improvements, mainly due to increases in spatial resolution and more sophisticated representations of important processes, particularly those associated with rainfall production and convection. These improvements have increased forecast accuracy. Combining the advantages of nowcasting for short lead times with the benefits of NWP models for longer range forecasts results in the best of both worlds in terms of Quantitative Precipitation Forecasting (QPF) for hydrological applications, particularly real-time flood forecasting.

Hydrological models themselves have improved significantly in recent years due to improved computer performance and their ability to capture the physical processes on a higher spatial and temporal resolution. Models are now much better at taking advantage of the data products offered by nowcasting and NWP models. Computers and telemetry technology are also much quicker at processing information.

Despite recent advances, it is clear that QPF remains a difficult problem. This is in part because precipitation is not one of the primary variables of the models and is estimated indirectly, often using some sort of parameterization scheme. The appropriateness of the scheme depends on the ability of the model to diagnose correctly the dominant rainfall-producing process. However, to the extent that NWP is an initial value problem, further improvements should be made by more accurately specifying the initial atmospheric state at higher resolution and in more detail.

It is well understood that the Earth's atmosphere is a chaotic non-linear system, resulting in predictions of behaviour that are sensitive to initial conditions (Lorenz 1963). Small perturbations in initial conditions can lead to significant differences later. Indeed, the growth of these differences is what limits the time ahead for which the forecasts are useful. More recently it has been demonstrated that some indications about the

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uncertainty in the forecasts can be simulated by perturbing the initial conditions of the NWP models being used by amounts that represent the uncertainty in the input data and running the forecast many times. The hope is that the distribution of the ensemble of modelled outputs gives a measure of the uncertainty in the forecast. For the purposes of this chapter, this approach will be termed 'ensemble methods'.

Ensembles of rainfall input for rainfall-runoff models generated by combining ensembles of rainfall extrapolated from remotely sensed data with ensembles of output from NWP models present an interesting and useful tool to flood forecasters and emergency responders. Instead of a deterministic process for developing the 'most likely' flood peak or emergency situation, the user now possesses a probability distribution of future hydrological conditions, which is much more useful for decision-making.

Hydrological Considerations

Despite the fact that lumped models often outperform distributed models, the latter are more often used for real-time flood forecasting because of their ability to deal more directly with spatially heterogeneous catchment characteristics and inputs, common in extreme events. However, recent studies have shown that careful parameterization and calibration can improve distributed model performance beyond what can be achieved used a lumped approach. The size of the catchment, the spatial resolution of the data available to describe the catchment, and the resolution of rainfall data used as input may dictate which modelling approach is most appropriate.

Quantitative Precipitation Forecasting from projections of radar and/or satellite data (nowcasting) has been used to estimate floods with greater lead time, but the accuracy of the forecasts diminishes dramatically for forecasts beyond 90 minutes. Mesoscale NWP models (often a high-resolution model nested within the grid of a lower resolution model and centred over the catchment of interest) have been used to provide better long-range quantitative precipitation forecast ensembles, but bring their own set of uncertainties. The source and magnitude of the uncertainty in NWPs is often poorly understood. Forecasts from NWP models are usually pre-processed to remove bias and often increased to match raingauge observations. NWP models are discussed in more detail in later sections.

There is a role for both nowcasting and NWP models in real-time flood forecasting, but it is important to understand in which situations combining the two is useful and improves the accuracy and confidence in the hydrological forecasts produced. Figure 10.1 shows the errors in quantitative rainfall forecast using various techniques and a combined system. The combined system consistently produced improved results. Where a hydrological application such as a small urban catchment requires a short lead time up to about 90 minutes, NWP models provide little value, and radar and real-time raingauge data are often used exclusively in these situations. By contrast, for larger catchments, intense but short-lived rainfall activity usually poses less concern than moderately heavy but longer duration rainfall over the entire region. This is particularly true where there are concerns regarding the reliability of levees or dam emergency spillways.

It makes sense to determine if a combined QPF is going to improve the accuracy and usefulness of a flood forecasting system before adopting this logistically complex and computationally intensive approach. Examining the response times of the catchment(s) is a logical first step. Particularly in mixed urban-rural settings, or where a large river flows through an urban setting and flooding may result from either overbank flow or insufficient capacity in the city's storm water drainage system, there may be a real advantage in merging several precipitation forecasting methods in a combined QPF. This is true even if separate hydrological models are being used.

Flood forecasters must attempt to minimize losses due to flooding while ensuring they do not lose the public trust through unnecessary warnings; knowledge of the uncertainty in the flood forecast is very important. This uncertainty

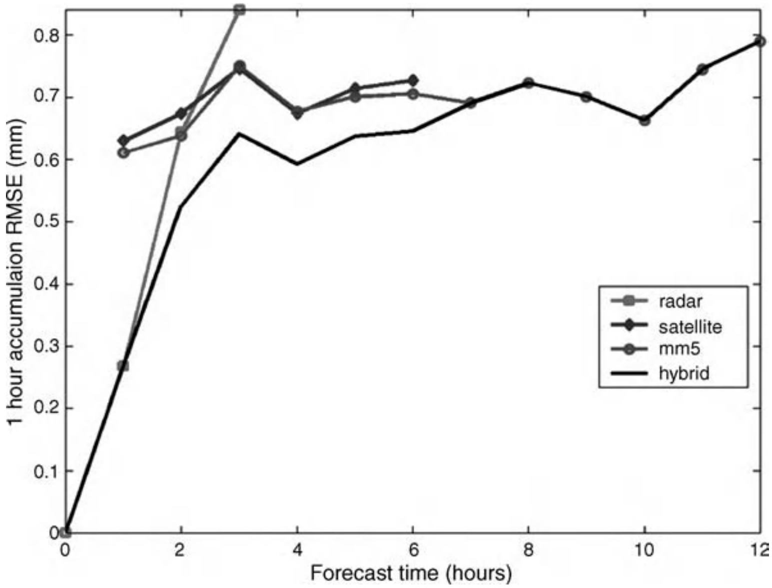

Fig. 10.1 Mean error associated with the forecasting of the amount of rainfall using radar, satellite, Numerical Weather Prediction (NWP) techniques and a hybrid scheme. From Smith and Austin (2000).

depends on many factors, including the degree of understanding of the physical characteristics of the watershed, the quality of the hydrological model, and the availability and accuracy of the information feeding the flood forecasting system. The single most important input is rainfall. The ability of the system to update predictions based on observed streamflow and revised precipitation estimates is also very important.

When the primary drivers of uncertainty are known, a probability distribution of flood flow from ensemble output can be developed. In this technique, variables affecting streamflow, including rainfall, are adjusted within a range of possibilities and the resulting hydrographs are then analysed statistically. This Monte Carlo approach has been used by hydrologists for many years. Both nowcasting and NWP methods offer the possibility of developing ensemble rainfall inputs for runoff simulation. Nowcasting algorithms can be developed to produce a realistic range of future storm direction, speed and growth. Similarly, NWP models are highly dependent on initial conditions and model parameterization. Small perturbations in these values can be used to develop a realistic range of forecasts. Combining the nowcasting ensembles with the NWP ensembles produces a set of QPFs that capture the range of uncertainty in the precipitation forecast. Routing this ensemble of QPFs through a rainfall-runoff model results in a probability distribution of future hydrological conditions, which is very useful for emergency management. The concept is shown schematically in Figure 10.2.

Developing numerous simulations of an already computationally expensive coupled meteorological-hydrological model is not realistic in many situations. Simplifying the NWP or rainfall-runoff model to reduce the CPU burden is an often successful but hardly intellectually pleasing approach. More often, researchers examine the probability distribution of Monte Carlo-generated QPF outputs and select individual rainfall forecasts that adequately represent this distribution. This can reduce the number of simulations (and hence computational time) by an order of magnitude. The process of developing a new QPF ensemble and routing it through the flood forecasting model is repeated as new observational data become available. The number of QPFs in the

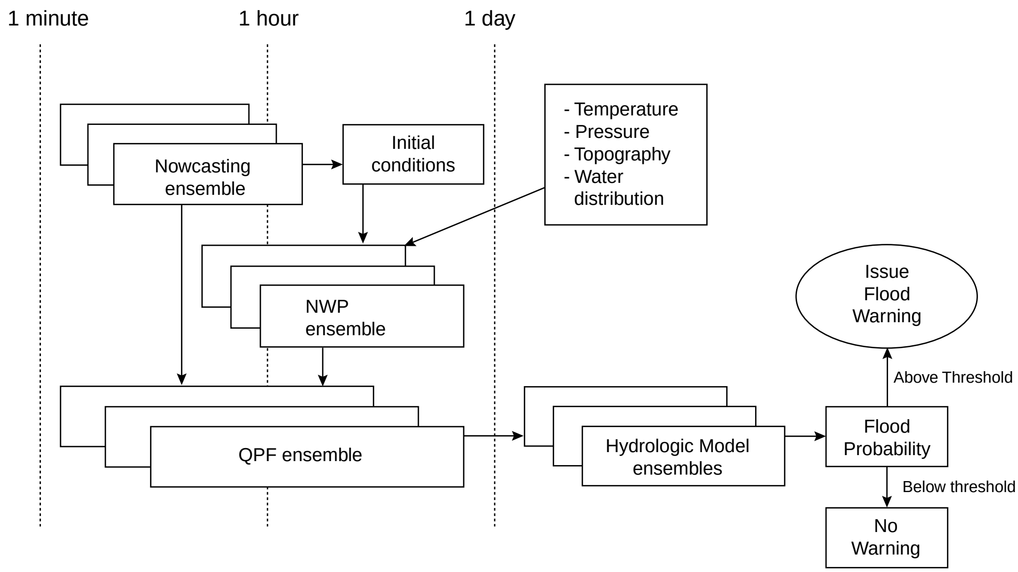

Fig. 10.2 Combining the advantages of nowcasting with Numerical Weather Prediction for improved flood decision support systems. QPF, quantitative precipitation forecasting.

ensemble is dictated by how long it takes to run the NWP and hydrological models. Beven *et al.* (2005) describe a process for constraining and propagating the uncertainty in a cascade of rainfall-runoff and flood routing models in an application to the River Severn in the UK.

Flood forecasting systems are used in areas where advanced flood warnings could prevent loss of life or reduce damages. The desired lead time corresponds to the time it takes to evacuate; the more time required, the more important it is to use a flood forecasting system that projects into the future with some degree of accuracy. In areas with poor drainage due to topography and/or poor flood conveyance or flood prevention infrastructure, there may be a very short period of time available between the formation of rainfall and surface inundation. As the rainfall event materializes and it becomes evident that flooding will occur, it becomes more important to estimate the spatial extent of inundation areas.

The traditional approach to flood forecasting is purely deterministic, where observed rainfall or a single rainfall forecast is input to a hydrological model and decisions are made based on the resultant flood hydrograph – specifically whether or not an area will flood and whether or not to evacuate or take other emergency action. Where the likely damage caused by a flood event is high, real-time streamgauges and water level monitoring equipment can be used to corroborate the model. However, usually the decision whether or not to take action is a judgment call by the emergency response team, who may or may not have good knowledge of the performance and reliability of the flood warning system. A flood forecast that has an associated probability of occurrence provides a framework for making more informed decisions. For example, an emergency management team may decide to call for the evacuation of part of a city if the probability of inundation is greater than 20%. By contrast, in a situation where a loss of life is likely, such as a dam or levee failure, evacuations may be called for if the probability of occurrence is much less.

The use of improved NWP models in hydrological applications is widespread, as is the use of nowcasting. Combined QPFs, where the advantages of both approaches is maximized, offers the hydrologist an improved approach to developing reliable short- and long-range forecasts within the same model input dataset. Developing an ensemble of precipitation forecasts enables the development of a probability distribution of the magnitude of the predicted flood, which in turn is a useful tool for those who respond to emergencies.

Meteorological Discussion

The Earth's atmosphere is characterized by fields of temperature and pressure that are relatively smooth in space and time when compared with the distribution of water, particularly water stored as cloud and rain. This is primarily because clouds and rain are produced by very non-linear processes involving condensation. The measurement of areal rainfall is therefore a difficult problem in view of the very high spatial heterogeneity of the rainfall pattern. The density of raingauges required to adequately represent the rainfall pattern is thus very high and depends on the meteorological processes that are giving rise to the rain. Radar remote sensing may be the only way to adequately represent the rainfall pattern associated with strongly convective systems (see Chapter 7).

The prediction of quantitative rainfall amounts is even more problematic, essentially due to these same non-linear processes. For the time domain of a few hours ahead, the extrapolation of radar or satellite imagery yields results that can be operationally useful. This process is generally known as 'nowcasting'. For longer time horizons, forecasts generated by solving the underlying dynamical equations need to be used. These models may be classified into 'global' and 'meso', depending on the scale at which they operate. All NWP models suffer from the difficulty that they too must attempt to deal with the chaotic behaviour of the real atmosphere, with rainfall probably being the most difficult parameter of the atmosphere to predict. The mesoscale models probably have a more difficult time since they are attempting to model very small-scale systems, which could well be convective in nature.

It is very important to be realistic about the difficulties NWP is going to have in representing small-scale precipitation events, such as that illustrated in Figure 10.3. Weather radar imagery of a small band of thunderstorms in New Zealand is

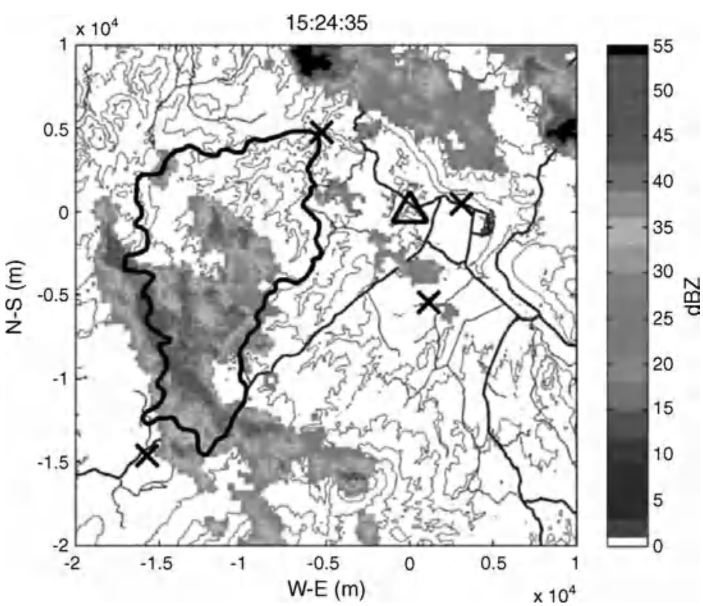

Fig. 10.3 Radar imagery of shower passing over a small catchment (outline solid black). The raingauge network was unable to sample spatial variability of the hydrometeor in this event. The radar is located at 0,0 (Δ). Terrain contours and roads have also been shown. (See the colour version of this figure in Colour Plate section.)

shown, which is interesting in that it was not represented in any of the national or local NWP models and did not register in any raingauge. Nevertheless the event produced significant surface flooding in the town of Tokoroa.

The operational solution to these difficulties is to develop a coupled system for QPF that is driven from the large scale by a global NWP model and from the small scale by remote-sensed data from radar and satellites via a nowcasting system. Then, of course, this system needs to be linked to an appropriate hydrological model to convert the QPF to a flood forecast.

Hydrometeorology

It is clear that water in the form of rain, ice and clouds is a major factor in the energy balance of the global atmosphere. This is because water vapour is by far the major greenhouse gas in the Earth's atmosphere and also because of the huge effects clouds have on the global radiation budget. The recognition of the intimate relationship between the global energy balance and global water cycle culminated in the establishment of the international Global Energy and Water Cycle Experiment (GEWEX) programme, which is dedicated to providing a better understanding of the processes underpinning this linkage (Lawford et al. 2004). Although NWP is modelling a complex and chaotic set of processes, it is still in essence a classical initial value problem. In spite of this, NWP models are often initialized with no clouds or rain. The models 'spin up' and subsequently generate their own clouds and rainfall patterns. Given the intimate relationship between the energy of the atmosphere at any time and the amount of liquid water and vapour it would seem highly desirable to include cloud and rainfall distributions in the data assimilated into NWP models. It is therefore to be expected that assimilation of high-resolution cloud water content and rainfall data from satellites and rain radars should result in improvements in the prediction of rain and severe weather in particular. In fact this has been demonstrated by many researchers, including Lin *et al*. (2005) and Mirza *et al*. (2008).

The estimation of rainfall distributions by radar has a long history, described in Atlas (1990). Its effective use, particularly in urban situations, has a shorter history but has become relatively well established in many places (see, e.g., Chapter 7 in this volume, and Austin and Austin, 1974). The estimation of rainfall amounts from satellite images has been attempted from the early days of meteorological satellites and the early work is reviewed in Barrett and Martin (1981). Satellite rainfall data are clearly required because even for a small catchment the air masses that subsequently produce the rainfall are very distant from the catchment hours or days earlier. Using visible and infrared (IR) geostationary data for the initial situation is clearly more realistic than assuming no initial rain at all (Lovejoy and Austin 1978). More recently, radars and microwave radiometers carried by satellites now offer better rainfall pattern measurements over large portions of the globe.

Estimates of global cloud water content based on microwave radiometers at a resolution of about 50 km are now available several times each day from the National Oceanic and Atmospheric Administration (NOAA). These estimates depend on the different radiative properties of the ground, water vapour in the atmosphere, and ice and water clouds. By using measurements in many different frequencies in the microwave part of the spectrum, it is in principle possible to separate the different components and thus estimate the path-integrated rain, water vapour, water and ice clouds. There are, however, some difficulties remaining, described in Horvath and Gentemann (2007) and elsewhere, and this approach works much better over ocean than over land. Higher-resolution images of clouds are also available from the Moderate Resolution Imaging Spectroradiometer (MODIS) and Multi-angle Imaging SpectroRadiometer (MISR) instruments, and some agreement is shown between their high-resolution results and the lower-resolution microwave results for large clouds. Difficulties remain in regions of partial cloud cover (Horvath and Davies 2007). This is clearly an important area of active research that will inevitably lead to improved resolution and accuracy of global mapping of cloud properties. Assimilation of this additional information into NWP models, provided that it is of sufficient quality, will improve predictions of the NWP models, particularly for QPF.

Nowcasting

As indicated earlier, at the convective scale of a kilometre or so the only practical way to obtain the precipitation pattern is by weather radar. The extrapolation of radar or satellite remote-sensed rainfall and cloud patterns to make QPF predictions a few hours ahead – nowcasting (Browning 1982) – has been used operationally for years, particularly for thunderstorm hazards at airports but also for urban and flash flooding. Various attempts have been made to combine nowcasting with NWP to take advantage of the superior predictive capability of nowcasting for the first couple of hours while retaining the guidance afforded by the NWP on the longer timescale. The most successful early attempt was the UKMetO Frontiers System (Browning 1982). More recently the Australian Bureau of Meteorology System, STEPS, has demonstrated some success (Bowler *et al*. 2006). In the latter system the spatial resolution of the output is reduced the further ahead is the forecast time, recognizing the impossibility of predicting small-scale features far into the future. Researchers are actively involved in looking for ways to statistically downscale the low-resolution forecasts to give an ensemble of QPF outcomes for hydrological models.

Numerical weather prediction (NWP)

Numerical Weather Prediction (NWP) can be considered a boundary and initial value problem – in order to make forecasts the starting point needs to be specified. Various NWPs can then be used to propagate the initial conditions (analysis) forward in time to generate a forecast.

The models are all similar in that they solve formulations of dynamic equations associated with the conservation of atmospheric mass, thermodynamic energy and momentum. In some models, vertical momentum considerations are simplified by making use of the hydrostatic approximation. More advanced non-hydrostatic models solve this explicitly. The direct treatment of vertical momentum turns out to be important on small scales where convection and updrafts need to be resolved by the model. At large scales these processes can be included in so-called subgrid parameterizations. Since these processes are the ones associated with clouds and precipitation their treatment can be particularly important for hydrological applications.

The high-resolution cloud and rain information discussed in the previous section needs to be assimilated into the NWP model, either as initial data or by 'nudging' a continuously functioning operational model. Whilst the estimation of initial precipitation and cloud fields from the NWP model is a relatively straightforward process, although not a particularly accurate one, the introduction of the cloud and rainfall information into NWP models is not. The models carry water as fields of cloud liquid, frozen water and water vapour, and for practical purposes treat rainfall as a diagnostic rather than a prognostic parameter. Analysis methods have yet to be fully established to adjust the moisture parameters and convergence fields such that the observed rain patterns are generated by the model at the initialization time. This is a non-trivial task.

Nowadays, 3D-VAR (three dimensional variational) (e.g. Lorenc *et al*. 2000) and 4D-VAR (e.g. Rawlins *et al*. 2007) methods are considered to be state of the art for the assimilation of observed data into the initial conditions of NWPs. In the nD-VAR assimilation systems, differences between observations, the model and background fields from a previous model run, and a new analysis are minimized in a least squares sense, weighted by estimates of observational and model error. Observations might include radiosonde soundings, measurements from ground stations, satellite images or radar data. Such variational methods have now largely superseded older methods such as nudging and diabatic or physical initialization.

Some meteorological observations can be compared almost directly to prognostic model variables. Wind, for example, is treated explicitly in NWP models, so a comparison between model wind and anemometer readings requires only a simple observational operator to account for interpolation from model space to a point measurement.

The treatment of precipitation-related measurements is somewhat more troublesome, as more complicated models are required to compare observations and prognostic model variables. Special care must be taken with the conditional on/off thresholds common to precipitation microphysical models, which can be difficult to represent as linear approximations.

Assimilation of large fields of observational data in the presence of clouds and precipitation data via VAR is a complex process. Inferring details of clouds and precipitation from radiance measurements is complicated because the exact propagation of radiation is highly dependent on the spatial distribution and micro-physical properties of the hydrometeors. The large number of parameters contributing to the measurement results in a poorly constrained problem, with errors that are difficult to determine. Often rain or cloud-affected satellite radiance data are excluded from operational assimilation schemes for this reason (Errico, Bauer and Mahfouf 2007), although in the long term this is not a tenable position to take given the high percentage of the globe that is covered by clouds at any time. There is much research currently underway in this area.

Rain radar data are also problematic to assimilate due to the inherent discontinuity in forward modelling from precipitation (or more correctly reflectivity) to humidity and temperature. Again significant difficulties are associated with the division between precipitation/no precipitation situations. This discontinuity can result in bimodal probability distributions (Errico *et al.* 2000), which are difficult to deal with in minimization schemes.

Nonetheless, case studies at various forecast centres have found that variational assimilation of radar reflectivity and/or radial velocity data results in improvement to model fields (Lopez and Bauer 2007; (Xiao and Sun 2007); (Rihan et al. 2008); (Pu, Li and Sun 2009); (Xiao et al. 2009).

Alternatively, moisture observations can be treated without VAR to generate pseudo-observations. The UK Met Office runs the Moisture Observation Pre-processing Syste (MOPS) operationally, which treats moisture observations before ingestion. The system combines IR satellite, rain radar and surface observations with the background model state to generate a distribution of humidity pseudo-observations (MacPherson *et al.* 1996) that can then be assimilated more readily against model humidity parameters. Radar observations are included in operational model runs via the Latent Heat Nudging (LHN) method (Bell 2009) (Dixon et al. 2009) described by Jones and MacPherson (1996). LHN involves modifying the latent heat in the model by the ratio of model to radar rain rate.

Regardless of the scheme used to assimilate rain and cloud observations into a model as initial conditions, the predictive power of this system will need to be tested. Comparison runs of the model with and without cloud and rain assimilation need to be performed. In this way the impact of changes to the initial conditions on the forecasts can thus be assessed by comparing the model outputs to observations. Further tests are then undertaken to characterize the sensitivity of the model to perturbations in the new initial conditions. This could be achieved by performing runs with subtly different rain/cloud schemes or by perturbing other model fields (Hohenegger and Schar 2007).

Preliminary results suggest that if the initialization is attempted by simply increasing the humidity field the extra moisture 'rains out' relatively quickly, leading to only short-term effects. If, however, the rainfall and cloud moisture fields are introduced by increasing the updrafts in the model, resulting in increased surface convergence and divergence aloft, then the model can move to a higher energy and greater rainfall state.

There has been some success reported with the addition of water vapour information to the initial conditions. Benedetti *et al.* (2005) assimilated radar reflectivity to adjust the humidity and temperature fields and observed an improvement in model skill, with the increases in moisture propagating to change the system dynamics. Water vapour has also been integrated based on microwave radiometer observations, and resulted in improved model predictions. A great deal of research is currently underway in this area and it is probably reasonable to expect that it will result in significant improvements in QPF in the next decade.

At the mesoscale, attempts to initialize NWP models with nowcast rainfall and cloud patterns are underway.

Ensembles

Since both the atmosphere and the hydrological response of the catchment contain highly nonlinear processes it is not plausible that a single deterministic representation of either QPF or hydrological output is likely to yield accurate results on all occasions. Thus there has been a growing interest in attempting to predict the distribution of QPF rainfall amounts as well as the distribution of possible hydrological responses. There are a variety of ways of trying to achieve this, which have only relatively recently become widely available because of the large demand such procedures usually place on computer resources. As discussed earlier, the initialization data for the NWP models are usually temperature, pressure, wind and humidity, historically derived primarily from radiosonde balloon ascents and more recently supplemented by satellite data, which are assimilated into the current cycle of the NWP process. A frequently used strategy is to randomly perturb the new input data by amounts comparable with the likely measurement errors and to then re-run the model. This process, known as Ensemble Forecasting, can then be repeated as many times as computer resources allow. Traditionally, operational ensembles have focused on synoptic scale baroclinic instability, which involves models not suitable for making predictions of rainfall on catchment scales. More recently, increasing computing power has seen the advent of regional ensembles such as the UK Met Office MOGREPS (Bowler et al. 2008). The resulting ensemble of outputs gives distributions for the predicted QPF, which can then be used to run many hydrological simulations, thus ultimately yielding a distribution of possible flow rates or river stages.

However, whilst this conceptually attractive procedure sometimes gives reasonable results for the likely distributions of the primary variables, much of the variability tends to be confined to the wind fields, and hence the QPF range is often smaller than actually observed, and in particular may not include the sort of extreme events likely to cause floods. To address this issue there have been attempts to consider a range of rainfall-producing processes in the NWP models in an attempt to capture the range of possible dominant cloud and microphysical processes that are parameterized in the NWP model. This approach is sometimes described as a 'physics ensemble' technique and can be accomplished by either using a suite of different NWPs or applying stochastic perturbations to parameters within one model's microphysics.

An alternative strategy is the so-called 'perturbation breeding' approach (Toth and Kalnay 1997) where the input data are perturbed everywhere and the NWP model is run for some equivalent time. Regions where large increases in the rainfall occur are then identified and the error field renormalized. The process is repeated to maximize the magnitude of the extreme event again. The idea is to determine the most exteme event that could actually happen from the modelled physics of the system. It is possible that this idea will make a useful contribution to extreme precipitation and flood frequency analysis.

Meteorological scale and process problems

For the purpose of considering the role and type of strategy required for effective flood warning, it is useful to divide meteorological situations into two distinct types of events: those that are forced from the large scale (e.g. fronts and cyclones) and those triggered or forced from the small scale (e.g. air mass thunderstorms and small-scale orographic effects). Most NWP models are best suited for making forecasts in the first category, where the model can take a large-scale view and effectively use the large-scale fluid dynamical outcomes to predict the likely movement and development of a front, for example. Thus the model is likely to get the arrival time of the rain from a front with good accuracy several hours, or even days ahead, but the detailed patterns in the QPF are likely to be quite wrong since there may be embedded convection or local small-scale eddies not resolved by the NWP model.

Orographic effects

A particularly problematic area for QPFs occurs in mountainous regions where the interaction between the large-scale wind patterns and the high-resolution topography can result in a number of significant highly non-linear behaviours, including the triggering of small-scale effects such as thunderstorms (Austin and Dirks 2005). In many countries the main flood hazards are flash floods in steep terrain. In these situations, the meteorology is likely difficult to model and the hydrology complex.

In New Zealand, for example, nearly all of the heavy rainfalls come either from subtropical cyclones encountering mountainous terrain or from strong onshore winds interacting with the Southern Alps. In both cases there is the need to allow high-resolution terrain to interact with large-scale flows. It is found that the NWP models give improved results as the resolution of the terrain and model are increased, leading to some hope for better flash flood forecasts in mountainous areas.

Important Research Questions

Whilst the idea of driving hydrological flood prediction models from the QPFs generated by atmospheric NWP models is entirely obvious and logical, there are serious questions about the ability of the NWP systems to represent and predict rainfall amounts with sufficient accuracy, particularly for real-time flood forecasting. Applications to climate studies may be less problematic. Many questions remain, however, including:

- Will the additional assimilation of water in the form of clouds and rain into NWP models improve their QPF accuracy to make them credible to drive hydrological models?
- If so, what lead times are achievable as a function of meteorological system type and scale of hydrological problem?
- What sort of hydrological problem lends itself to combined nowcasting/NWP/hydrological modelling?

Prospects/Conclusions

There are currently great opportunities for real improvements in the way meteorological forecasts are used for flood warning systems. Recent developments include drastic improvements in the resolution, physics and data assimilation of quantitative rainfall amounts in NWP models. Similarly the performance of distributed rainfall-runoff models suitable for flood prediction has notably improved. Moreover, large computing facilities have declined markedly in cost, including inexpensive parallel processing clusters, making multiple runs of the meteorological and hydrological models operationally feasible. This in turn opens up the possibility of working with ensembles of model outputs, thus giving information about the likely range and also perhaps extrema of flooding events. Whilst the interpretation of such results is not trivial, it is expected that they will be of much greater value for those charged with giving flood warnings.

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Part 4
Flood Modelling and Mitigation

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11 Data Utilization in Flood Inundation Modelling

DAVID C. MASON, GUY J-P. SCHUMANN
AND PAUL D. BATES

Introduction

Flood inundation models are a major tool for mitigating the effects of flooding. They provide predictions of flood extent and depth that are used in the development of spatially accurate hazard maps. These allow the assessment of risk to life and property in the floodplain, and the prioritization of either the maintenance of existing flood defences or the construction of new ones.

There have been significant advances in flood inundation modelling over the past decade. Progress has been made in the understanding of the processes controlling runoff and flood wave propagation, in simulation techniques, in low-cost high-power computing, in uncertainty handling, and in the provision of new data sources.

One of the main drivers for this advancement has been the veritable explosion of data that have become available to parameterize and validate the models. The acquisition of the vast majority of these new data has been made possible by developments in the field of remote sensing (Smith *et al*. 2006; Schumann *et al*. 2009). Remote sensing, from both satellites and aircraft, allows the rapid collection of spatially distributed data over large areas, and reduces the need for costly ground survey. The two-dimensional synoptic nature of remotely sensed data has allowed the growth of two- and higher-dimensional inundation models, which require 2D data for their parameterization and validation. The situation has moved from a scenario in which there were often too few data for sensible modelling to proceed, to one in which (with some important exceptions) it can be difficult to make full use of all the available data in the modelling process.

This chapter reviews the use of data in present-day flood inundation modelling. It takes the approach of first eliciting the data requirements of inundation modellers, and then considering the extent to which these requirements can be met by existing data sources. The discussion of the data sources begins by examining the use of data for model parameterization. This includes a comparison of the main methods for generating Digital Terrain Models (DTMs) of the floodplain and channel for use as model bathymetry, including airborne scanning laser altimetry (Light Detection and Ranging: LIDAR) and airborne Interferometric Synthetic Aperture Radar (InSAR). Filtering algorithms for LIDAR data are reviewed, as are the use of remotely sensed data for distributed floodplain friction measurement and the problems of integrating LIDAR data into an inundation model. A detailed discussion follows on the use of remotely sensed flood extent and water stage measurement for model calibration, validation and assimilation. Flood extent mapping from a variety of sensors is considered, and the advantages of active microwave systems highlighted. Remote sensing of water stage, both directly by satellite altimeters and InSAR and indirectly by intersecting flood

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extents with DTMs, is discussed. The integration of these observations into the models involves quantification of model performance based on flood extent and water levels, and consideration of how model performance measures can be used to develop measures of uncertainty via flood inundation uncertainty maps. The assimilation of water stage measurements into inundation models is also discussed. The article concludes by considering possible future research directions that aim to reduce shortfalls in the capability of current data sources to meet modellers' requirements.

Data Requirements for Flood Inundation Modelling

The data requirements of flood inundation models have been reviewed by Smith *et al*. (2006). They fall into four distinct categories:

1 topographic data of the channel and floodplain to act as model bathymetry;

2 time series of bulk flow rates and stage data to provide model input and output boundary conditions;

3 roughness coefficients for channel and floodplain, which may be spatially distributed;

4 data for model calibration, validation and assimilation.

The basic topographic data requirement is for a high-quality Digital Terrain Model (DTM) representing the ground surface with surface objects removed. For rural floodplain modelling, modellers require that the DTM has vertical accuracy of about 0.5 m and a spatial resolution of at least 10 m (Ramsbottom and Wicks 2003). Whilst this level of accuracy and spatial scale is insufficient to represent the microtopography of relict channels and drainage ditches existing on the floodplain that control its initial wetting, at higher flood depths inundation is controlled mainly by the larger scale valley morphology, and detailed knowledge of the microtopography becomes less critical (Horritt and Bates 2001). Important exceptions are features such as embankments and levees controlling overbank flow, for which a higher accuracy and spatial scale are required (~10 cm vertical accuracy and 2 m spatial resolution) (Smith *et al*. 2006). This also applies to the topography of the river channels themselves. However, for modelling over urban floodplains knowledge of the microtopography over large areas becomes much more important, and a vertical accuracy of 5 cm with a spatial resolution of 0.5 m is needed to resolve gaps between buildings (Smith *et al*. 2006). Modellers also require a variety of features present on the ground surface to be measured and retained as separate GIS (geographical information system) layers to be used for tasks such as determining distributed floodplain roughness coefficients. Layers of particular interest include buildings, vegetation, embankments, bridges, culverts and hedges. One important use for these is for adding to the DTM critical features influencing flow paths during flooding, such as buildings, hedges and walls. A further use is the identification and removal of false blockages to flows that may be present in the DTM, such as bridges and culverts.

Flood inundation models also require discharge and stage data to provide model boundary conditions. The data are usually acquired from gauging stations spaced 10–60 km apart on the river network, which provide input to flood warning systems. Modellers ideally require gauged flow rates to be accurate to 5% for all flow rates, with all significant tributaries in a catchment gauged. However, problems with the rating curve extrapolation to high flows and gauge bypassing might mean that discharge measurement errors can be much higher than this acceptable value during floods. At such times gauged flow rates are likely only to be accurate to 10% at best, and at many sites errors of 20% will be much more common. At a few sites where the gauge installation is significantly bypassed at high flow, errors may even be as large as 50%.

Estimates of bottom roughness coefficients in the channel and floodplain are also required. The role of these coefficients is to parameterize those energy losses not represented explicitly in the model equations. In practice, they are usually estimated by calibration, which often results in them compensating for model structural and input flow errors. As a result, it can be difficult to disentangle the contribution due to friction from that attributable to compensation. The simplest method of calibration is to calibrate using two separate global coefficients, one for the channel and the other for the floodplain. However, ideally friction data need to reflect the spatial variability of friction that is actually present in the channel and floodplain, and be calculable explicitly from physical or biological variables.

A final requirement is for suitable data for model calibration, validation and assimilation. If a model can be successfully validated using independent data, this gives confidence in its predictions for future events of similar magnitude under similar conditions. Until recently, validation data for hydraulic models consisted mainly of bulk flow measurements taken at a small number of points in the model domain, often including the catchment outlet. However, the comparison of spatially distributed model output with only a small number of observations met with only mixed success (Lane *et al*. 1999). The 2-D nature of modern distributed models requires spatially distributed observational data at a scale commensurate with model predictions for successful validation. The observations may be synoptic maps of inundation extent, water depth or flow velocity. If sequences of such observations can be acquired over the course of a flood event, this allows the possibility of applying data assimilation techniques to further improve model predictions.

Use of Data for Model Parameterization

This section discusses the extent to which the data requirements of the previous section can be met by existing data sources, including any shortfalls that exist.

Methods of Digital Terrain Model Generation

The data contained in a DTM of the floodplain and channel form the primary data requirement for the parameterization of a flood inundation model. Several methods exist for the generation of DTMs suitable for flood modelling. Smith *et al*. (2006) have provided an excellent review of these, together with their advantages and disadvantages for flood inundation modelling, and this is summarized below. The choice of a suitable model in any given situation will depend upon a number of factors, including the vertical accuracy, spatial resolution and spatial extent required, the modelling objectives and any cost limitations. Many air- and space-borne sensors generate a Digital Surface Model (DSM), a representation of a surface including features above ground level such as vegetation and buildings. A DTM (also called a Digital Elevation Model (DEM)) is normally created by stripping off above-ground features in the DSM to produce a 'bald-earth' model.

Cartography

A DTM can be produced by digitizing contour lines and spot heights from a cartographic map of the area at a suitable scale, then interpolating the digitized data to a suitable grid (Kennie and Petrie 1990). The product generated is a DTM since ground heights are digitized. While the method is relatively economical, contour information is generally sparse in floodplains because of their low slope, which limits the accuracy of the DTM in these areas.

An important example of such a DTM for the UK is the Ordnance Survey Landform Profile Plus DTM, which is of sufficiently great height accuracy and spatial resolution to be useful for flood risk modelling (www.ordnancesurvey.co.uk/oswebsite/products/landformprofileplus). This has been developed from the Landform Profile dataset (which was generated from 1:10,000 contour maps and covers the entire UK (Holland 2002)) by combining it with LIDAR and photogrammetric data. The Profile Plus DTM has a vertical accuracy and spatial resolution that depend on land cover type, being ± 0.5 m on a 2-m grid in selected urban and floodplain areas, ± 1.0 m on a 5-m grid in rural areas, and ± 2.5 m on a 10-m grid in mountain and moorland areas.

Ground survey

Elevations can be measured directly in the field using total stations or the Global Positional

System (GPS). The spot heights measured have to be interpolated onto a grid to produce a DTM. While these techniques provide the greatest accuracies currently achievable, they require lengthy fieldwork, making them more suitable for providing validation data for other techniques or for filling gaps in data than for DTM generation over large areas. Total stations are electronic theodolites with distance measurement capabilities, which can position points to better than ±0.5 cm (Kavanagh 2003). GPS is a system that provides continuous all-weather positioning on the Earth's surface using a constellation of more than 24 satellites transmitting continuous microwave signals (see, e.g., Hoffman-Wellenhof *et al.* 1994). The two main observing modes used in surveying are differential static positioning and kinematic positioning, each of which require as a minimum a base and a roving receiver. Static positioning can achieve a positional accuracy of ±2 cm, whereas kinematic positioning, requiring less observation time, can achieve ±5 cm.

Digital aerial photogrammetry

Digital Terrain Models can be produced using stereo-photogrammetry applied to overlapping pairs of aerial photographs (Wolf and Dewitt 2000). Photographs are usually acquired in strips with adjacent photographs having 60% overlap in the flight direction and 20–30% overlap perpendicular to this. They are then digitized using a photogrammetric scanner. Coordinates in the camera's image coordinate system are defined knowing the imaging characteristics of the camera and the location of a set of fiducial marks. A relationship between the image space coordinates and the ground space coordinates is determined in modern aircraft systems using the onboard GPS to determine positions and the inertial navigation system (INS) to determine orientations. In order to generate ground elevations from a stereo-pair, corresponding image points in the overlapping area of the pair must be determined. While this image matching can be performed semi-automatically, automatic area-, feature- or relation-based matching is less time-consuming. The 3-D ground space coordinates of points can then be determined, and interpolated onto a regular grid. The model formed is essentially a DSM. Semi-automatic techniques are used to remove blunders. The accuracy achieved depends on the scale of the photography and the skill of the operator.

Photogrammetric techniques have been used in the development of the Ordnance Survey (OS) Landform Profile Plus DTM. There is extensive aerial photography of the UK, and UK Perspectives has created a DTM of the UK using photogrammetry applied to 1:10,000- and 1:25,000-scale imagery. This has an approximate vertical accuracy of ±1 m and a 10-m grid spacing.

Interferometric synthetic aperture radar (InSAR)

A DSM can be generated using InSAR, which uses two side-looking antennae on board a satellite or aircraft separated by a known baseline to image the terrain (Goldstein *et al.* 1988; Madsen *et al.* 1991). Two main configurations exist: repeat pass interferometry, where the data are acquired from two passes of a (usually satellite) sensor in similar orbits; and single pass interferometry, where the data are acquired in a single pass using two antennae separated by a fixed baseline on the same platform, which to date has been an aircraft or the Space Shuttle. The height of a point can be determined by trigonometry, using knowledge of the locations and orientations of the two antennae (from GPS and each sensor's INS), their baseline separation, and the path difference between the signals from each antenna. The surface elevation measured for a pixel may consist of a combined signal from different scatterers in the pixel. For pixels containing vegetation, volume scattering will occur and there will be some penetration into the canopy, so that the height measured will not be that of the first surface. Other limitations are that performance can degrade in urban areas due to bright targets and shadow, and that artefacts may appear in the DSM due to atmospheric propagation and hilly terrain. However, InSAR is all-weather and day-night, and large areas can be mapped rapidly.

A near-global high-resolution DSM of the Earth's topography was acquired using InSAR by the Shuttle Radar Topography Mission (SRTM) on board Space Shuttle Endeavour in February 2000. The SRTM was equipped with two radar antennae separated by 60 m, and collected data over about 80% of the Earth's land surface, between latitudes 60°N and 54°S. The vertical accuracy is about ±16 m, with a spatial resolution of 30 m in the USA and 90 m in all other areas (Smith and Sandwell 2003).

An InSAR DSM of the UK was produced using repeat pass InSAR techniques applied to Earth Resources Satellite-2 (ERS-2) satellite data in the LandMap project (Muller 2000). This has a height standard deviation of ±11 m and a spatial resolution of 25 m.

The main airborne InSAR is the InterMap STAR-3i. This is a single-pass across-track X-band SAR interferometer on board a Learjet 36, with the two antennae separated by a baseline of 1 m. In the NextMap Britain project in 2002–03, an accurate high-resolution DSM of the whole of Britain was built up containing over 8 billion elevation points. This meant that for the first time there was a national height database with height accuracies better than ±1 m and spatial resolutions of 5 m (10 m) in urban(rural) areas (www.intermap.com). Using in-house software, Intermap is able to filter the DSM to strip away features such as trees and buildings to generate a bare-earth DTM.

Light Detection and Ranging (LIDAR)

Light Detection and Ranging (LIDAR) is an airborne laser mapping technique that produces highly accurate and dense elevation data suitable for flood modelling (Wehr and Lohr 1999; Flood 2001). A LIDAR system uses a laser scanner mounted on an aircraft or helicopter platform (Fig. 11.1). Pulses from the laser are directed towards the Earth's surface, where they reflect off features back towards the platform. Knowing the round-trip time of the pulse and the velocity of light, the distance between the laser and the ground feature can be calculated. The instantaneous position and orientation of the laser are known using the GPS and INS systems on board the platform. Using additional information on the

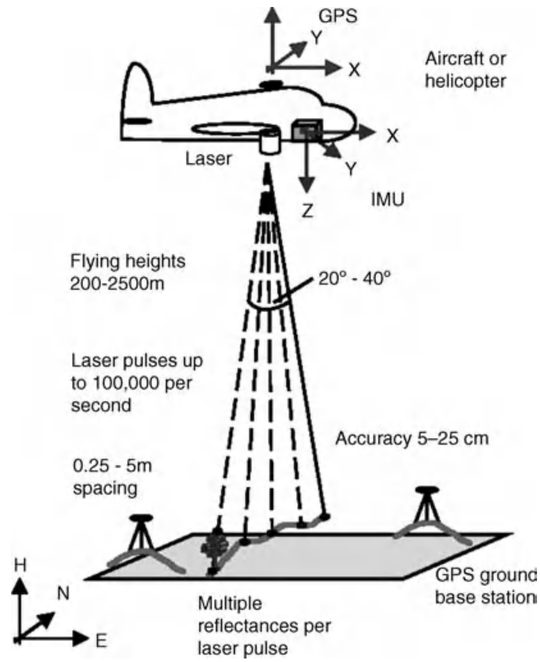

Fig. 11.1 Typical LIDAR (Light Detection and Ranging) system and its main components. GPS, global positioning system. After Smith *et al.* (2006).

scan angle and GPS base station, the 3-D position of the ground feature can be calculated in the GPS coordinate system (WGS84) and then transformed to the local map projection. A high vertical accuracy of ±5–25 cm can be achieved. At typical flight speeds, platform altitudes and laser characteristics, terrain elevations can be collected at a density of at least one point every 0.25–5 m. The laser pulse may reflect from more than one part of a ground feature; for example, in vegetated areas the pulse may reflect from the top of the foliage and also from the ground below. Many LIDAR systems can collect both the first return (from the foliage) and the last return (from the ground), and in some systems it is possible to collect the complete reflected waveform. The intensity of the reflected pulse can also provide useful information about the surface feature being imaged.

High-resolution LIDAR data suitable for flood modelling are available for a number of selected floodplain and coastal areas of the UK.

A substantial amount of these data have and are being collected by the Environment Agency of England and Wales (EA). Flights are typically carried out during leaf-off periods with the system set to record the last returned pulse. The EA provides quality control by comparing the LIDAR heights on flat unvegetated surfaces with GPS observations, and can achieve discrepancies of less than ± 10 cm (EA 2005). However, note that DTM errors generally increase in regions of dense vegetation and/or steep slope, and can be especially significant at the boundaries between river channels and floodplains.

Sonar bathymetry

Methods of estimating river channel topography usually involve generating a series of height cross-sections along the channel using ground surveying techniques, then interpolating between the cross-sections. With the advent of more sophisticated modelling, there is a need for better estimates of channel topography, and one technique involves bathymetric measurement using sonar. This uses a vessel-mounted transducer to emit a pulse of sound towards the river bed and measure the elapsed time before the reflection is received. The depth of water under the vessel can be estimated knowing the velocity of sound in water. In the UK, the EA operates a wide-swath sonar bathymetry system designed to make it straightforward to merge bathymetry of the channel with LIDAR heights on the adjacent floodplain (Horritt *et al*. 2006).

Suitability of DTM generation techniques for flood modelling

The suitability of a DTM generation technique for flood modelling is largely governed by the heighting accuracy and level of spatial detail that can be captured. Table 11.1 gives a summary of the main merits and limitations of available DTM generation techniques, and Table 11.2 summarizes the main characteristics of the DTMs that are generally available in the UK. Many of the techniques described in Table 11.1 produce DTMs that are not suitable for the quality of flood modelling currently being undertaken. Smith *et al*. (2006) point out that recently in the UK it is largely the LIDAR and swath bathymetry data collected by the EA, and the InSAR data collected by InterMap that have been used to produce DTMs for flood modelling. In parts of the world where LIDAR data are not available, floods are larger than in the UK, or modelling requirements are less stringent, other data sources (e.g. SRTM data) have been used (Wilson *et al*. 2007). However, the discussion below focuses on DTMs produced using LIDAR.

Filtering algorithms for LIDAR data

Considerable processing is necessary to extract the DTM from the raw DSM. The basic problem in LIDAR post-processing is how to separate ground hits from hits on surface objects such as vegetation or buildings. Ground hits can be used to construct a DTM of the underlying ground surface, while surface object hits, taken in conjunction with ground hits, allow object heights to be determined. Many schemes have been developed to perform LIDAR post-processing. Most of these are concerned with the detection and recognition of buildings in urban areas (Maas and Vosselman 1999; Oude Elberink and Maas 2000), or the measurement of tree heights (Naesset 1997; Magnussen and Boudewyn 1998). Commercial software is also available for the removal of surface features. Gomes Pereira and Wicherson (1999) generated a DEM from dense LIDAR data for use in hydrodynamic modelling, after the removal of surface features by the data supplier. Another example is the system developed by the EA, which uses a combination of edge detection and the commercial TERRASCAN software to convert the DSM to a DTM (A. Duncan, personal communication). The system has been designed with flood modelling in mind, and, as well as the DTM, also produces other datasets for use in the subsequent modelling process, including buildings, taller vegetation (trees, hedges) and embankments. An example of the EA's hybrid filtering process is shown in Figure 11.2. False blockages to flow such as bridges and flyovers are removed from the

Table 11.1 Summary of merits and limitations of available Digital Terrain Modelling (DTM) techniques (after Smith *et al.* (2006)

Methods	Merits	Limitations
Cartography	Simple to generate if digital contours are available Economical for large areas	Highly dependent on the scale and quality of the base map Does not accurately characterize low-lying areas such as floodplains Influenced by the skill of operator digitizing the map
Ground surveying	Extremely accurate Total Stations can acquire elevations under canopy Provides measurements for filling in voids in other datasets	Expensive and time consuming to collect for large areas GPS does not provide reliable heights under canopy Access required to property for measurement of heights
Digital aerial photogrammetry	Proven and well-understood approach Potential for high accuracies in plan and height Provides an optical image for interpretation Relatively economical for surveys of large areas	Delay between acquisition of images and production of DTM Dependent on scale and quality of imagery Limitations in the automatic matching algorithm Manual measurements require an experienced observer
Interferometric Synthetic Aperture Radar (SAR)	Can 'see' through clouds and operate day or night Rapidly maps very large areas	Volume scattering in vegetated areas leads to poor coherence Performance can degrade in urban areas due to bright targets and shadows Artefacts in the DTM due to topography or atmospheric propagation
LIDAR	Potential for high accuracy Can generate DTM for surface with little or no texture Could measure vegetation height when set to record first and last pulse	May require a lot of flying time for extremely large areas Cannot operate in cloudy, rainy or windy conditions May require complementary data, such as photo, if interpretation of points is necessary

DTM, digital terrain model; GPS, global positional system; LIDAR, Light Detection and Ranging.

LIDAR data manually using an image processing package, and the resulting gaps interpolated, prior to DSM filtering.

Floodplain friction measurement

Remotely sensed data may be used to generate spatially distributed floodplain friction coefficients for use in 2-D inundation modelling. A standard method is to use two separate global static coefficients, one for the channel and the other for the floodplain, and to calibrate these by minimizing the difference between the observed and predicted flood extents. The remote-sensing approach has the advantage that it makes unnecessary the non-physical fitting of a global floodplain friction coefficient. Wilson and Atkinson (2007) estimated friction coefficients from floodplain land cover classification of Landsat Thematic Mapper (TM) imagery, and found that spatially distributed friction had an effect on the timing of flood inundation, though less effect on the predicted inundation extent.

Data from LIDAR may also be used for friction measurement. Most LIDAR DSM vegetation removal software ignores short vegetation less than 1 m or so high. However, even in an urban floodplain, a significant proportion of the land surface may be covered with this type of vegetation, and for floodplains experiencing relatively shallow inundation the resistance due to vegetation may dominate the boundary friction term. Mason *et al.* (2003) extended LIDAR vegetation height measurement to short vegetation using local

Table 11.2 Summary of characteristics of generally available digital terrain models (DTMs) (after Smith *et al.* (2006)

Available DTMs	Method of generation	Spatial resolution (m)	Vertical accuracy ± (m)	Formats	Coverage	Estimate of costs[a]	Organization responsible
LandMap Elevation Data	Repeat-pass spaceborne InSAR	25	Varies (10–100)	DSM	Entire UK	Free for academics via Edina	MIMAS www.landmap.ac.uk
SRTM	Single-pass spaceborne InSAR	90	~16	DSM	Global	Free for research	JPL at NASA www.nasa.org www.edina.com
Land Form Profile	Cartographic (Digitized Contours)	10	~2.5–5.0	DTM	Entire UK	Free for academics via Edina or £4.20 per 5 × 5 km	Ordnance Survey
Land Form Profile Plus	Photogrammetry LIDAR	2 5 10	0.5 1.0 2.5	DTM	Being rolled out on a needs basis	£61–575 per 5 × 5 km	Ordnance Survey
NextMap Britain	Airborne InSAR	5 5 or 10	0.5–1.0 0.7–1.0	DSM DTM	England, Wales and Scotland	From £40 per km^2 (or £1 per km^2 via CHEST; minimum £500)	Intermap, Getmapping or BlueSky
LIDAR	Airborne LIDAR	0.25–3.0	0.05–0.25	DSM/DTM	Selected low-lying and coastal areas	From £800 per 2 × 2 km	Environment Agency
UK Perspectives	Photogrammetry	10	1.0	DTM	England	£25.00 per km^2	Simmons Aerofilms Ltd or UK Perspectives

[a]At time of writing User Focused Measurable Outcome (UFMO) report, June 2006.
DSM, Digital Surface Model; InSAR, Interferometric Synthetic Aperture Radar; LIDAR, Light Detection and Ranging; SRTM, Shuttle Radar Topography Mission.

height texture, and investigated how the vegetation heights could be converted to friction factors at each node of a finite element model's mesh. A system of empirical equations that depended on vegetation height class was used to convert vegetation heights to Manning's *n* values. All the friction contributions from the vegetation height classes in the polygonal area surrounding each finite element node were averaged according to their areal proportion of the polygon.

This process has been taken further in rural floodplains by decomposing the model mesh to reflect floodplain vegetation features such as hedges and trees having different frictional properties to their surroundings, and significant floodplain topographic features having high height curvatures (Bates *et al.* 2003; Cobby *et al.* 2003). The advantage of this approach is that the friction assigned to a node can be made more representative of the land cover within the node, so that the

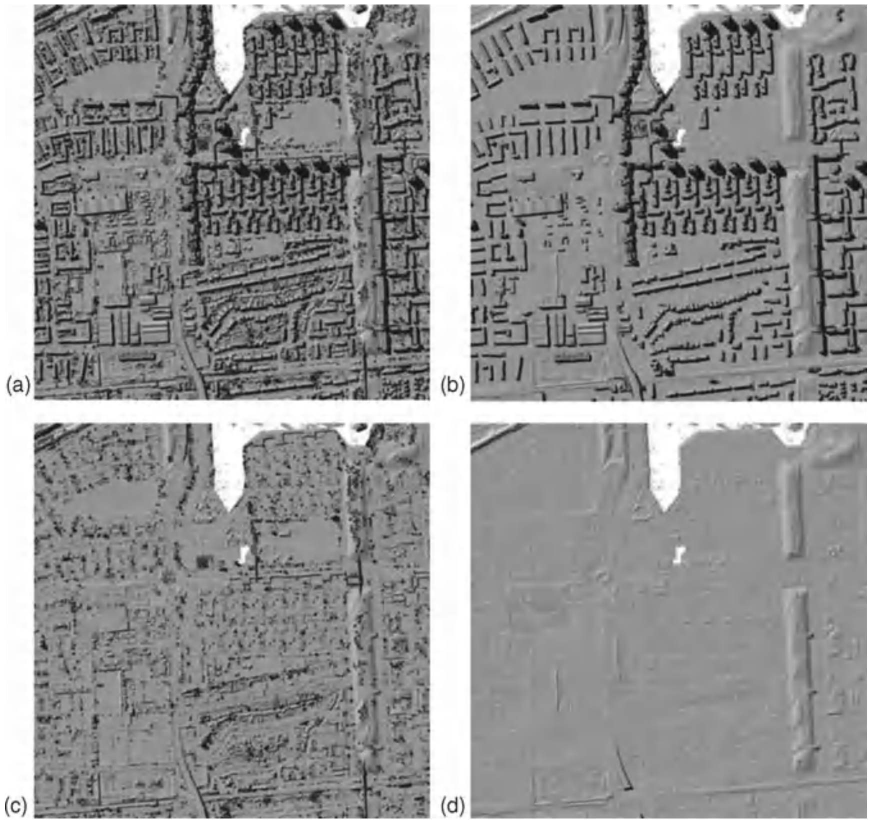

Fig. 11.2 Example of the Environment Agency Geomatricss Group's hybrid filtering. (a) Digital Surface Model; (b) Digital Terrain Model (DTM) with buildings; (c) DTM with vegetation; and (d) DTM without bridges. Reproduced with permission of Geomatrics Group, Environment Agency. (See the colour version of this figure in Colour Plate section.)

impact of zones of high friction but limited spatial extent (e.g. hedges) is not lost by averaging over a larger neighbourhood. The simulated hydraulics using the decomposed mesh gave a better representation of the observed flood extent than the traditional approach using a constant floodplain friction factor. The above technique has been extended for use in urban flood modelling using a LIDAR post-processor based on the fusion of LIDAR and digital map data (Mason *et al.* 2007a). The map data were used in conjunction with LIDAR data to identify different object types in urban areas, in particular buildings, roads and trees (Fig. 11.3).

Integrating LIDAR data into a flood inundation model

A problem with integrating LIDAR data as bathymetry into a flood inundation model is that the LIDAR data generally have a higher spatial resolution than the model grid. Marks and Bates (2000), who were the first to employ LIDAR as bathymetry in a 2-D model, coped with this by using the average of the four central LIDAR heights in a grid cell as the topographic height for the cell. Bates (2000) also used LIDAR in a subgrid parameterization in order to develop an improved wetting-drying algorithm for partially-wet grid

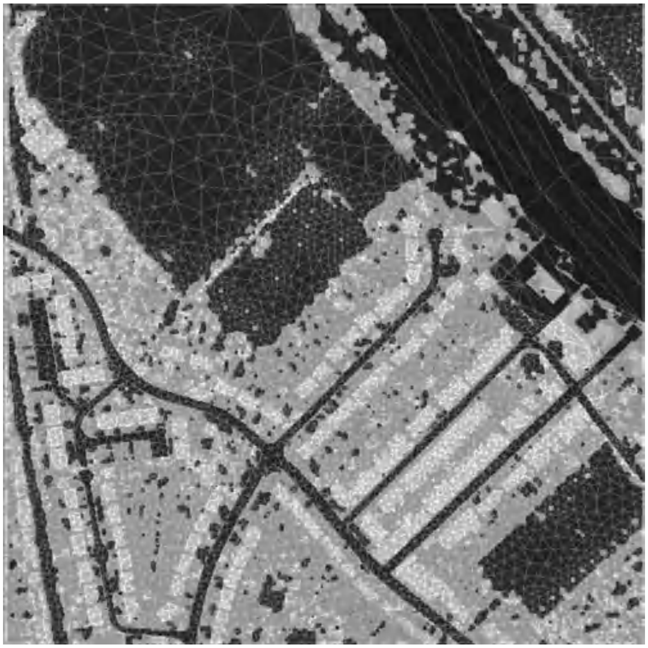

Fig. 11.3 Mesh constructed over vegetated urban area (red = mesh, blue = building/taller vegetation heights; a river is present in the northeast corner). After Mason *et al.* (2007a). (See the colour version of this figure in Colour Plate section.)

cells. If LIDAR data are averaged to represent DTM heights on a lower-resolution model grid (e.g. 1-m LIDAR data averaged to a 10-m model grid), care must be taken not to smooth out important topographic features of high spatial frequency such as embankments. Map data can be used to identify the embankments so that this detail can be preserved in the DTM (Bates *et al.* 2006).

In urban flood modelling studies using lower-resolution models where a grid cell may occupy several buildings, different approaches to the calculation of effective friction on the cell have been developed, based on object classification from LIDAR or map data. The first approach simply masks out cells that are more than 50% occupied by buildings, treating the edges of the masked cells as zero flux boundaries. The second uses a porosity approach, where the porosity of a cell is equal to the proportion unoccupied by buildings and therefore available for flow (Bates 2000; Defina 2000). Friction in the porous portion of the cell may then be assigned locally or globally.

The effect of errors in LIDAR DTMs on inundation predictions in urban areas has been considered in Neelz and Pender (2006) and Hunter *et al.* (2008). These studies concluded that uncertainty in friction parameterization is a more dominant factor than LIDAR topography error for typical problems. This is considered in more detail in the following chapter.

Use of Remotely Sensed Flood Extent and Water Stage Measurements for Model Calibration, Validation and Assimilation

Early launches of satellites and the availability of aerial photography allowed investigation of the potential to support flood monitoring from space. There have been notable studies on integrating data from these instruments with flood modelling since the late 1990s. A more recent consensus among space agencies to strengthen the support that satellites can offer has stimulated more research in this area, and significant progress has been achieved in recent years in fostering our understanding of the ways in which remote sensing can support or even advance flood modelling.

Flood extent mapping

Given the very high spatial resolution of the imagery, flood extent is derived from colour or panchromatic aerial photography by digitizing the boundaries at the contrasting land–water interface. The accuracy of the derived shoreline may vary from 10 to 100 m, depending largely on the skills of the photo interpreter, of which the georectification error is generally 5 m with ~10% chance of exceeding that error (Hughes *et al.* 2006).

In recent years, however, mapping flood area and extent from satellite images has clearly gained in popularity, mostly owing to their relatively low post-launch acquisition cost. Following a survey of hydrologists, Herschy *et al.* (1985) determined that the optimum resolution for floodplain mapping was 20 m, while that for flood extent mapping was 100 m (max. 10 m, min. 1 km) (Blyth 1997). Clearly, most currently available optical, thermal as well as active microwave sensors satisfy these requirements (Bates *et al.* 1997; Smith 1997; Marcus and Fonstad 2008; Schumann *et al.* 2009). Flood mapping with optical and thermal imagery has met with some success (Marcus and Fonstad 2008); however, the systematic application of such techniques is hampered by persistent cloud cover during floods, particularly in small to medium-sized catchments where floods often recede before weather conditions improve. Also, the inability to map flooding beneath vegetation canopies, as demonstrated by, for example, Hess *et al.* (1995, 2003) and Wilson *et al.* (2007) with radar imagery, limits the applicability of these sensors. Given these limitations for acquiring flood information routinely, flood detection and monitoring seems realistically only feasible with microwave (i.e. radar) remote sensing, as microwaves penetrate cloud cover and are reflected away from the sensor by smooth open water bodies.

The use of passive microwave systems over land surfaces is difficult given their large spatial resolutions of 20 to 100 km (Rees 2001). Interpretation of the wide range of materials with many different emissivities is thus rendered nearly impossible. Nevertheless, as the sensor is sensitive to changes in the dielectric constant, very large areas of water, for instance, can be detected (Sippel *et al.* 1998; Jin 1999) but their uncertainties may be large (Papa *et al.* 2006). Imagery from (active) SAR seems at present to be the only reliable source of information for monitoring floods on rivers <1 km in width. Although the operational use of SAR images for flood data retrieval is currently still limited by restricted temporal coverage (up to 35 days for some sensors), recent efforts on satellite constellations (e.g. COSMO-SkyMed) seem promising and should make space-borne SAR an indispensable tool for hydrological/hydraulic studies in the near future.

Many different SAR image-processing techniques exist to derive flood area and/or extent. They range from simple visual interpretation (e.g. Oberstadler *et al.* 1997) and image histogram threshold (Otsu 1979) or texture measures, to automatic classification algorithms (e.g. Hess *et al.* 1995; Bonn and Dixon 2005) or multi-temporal change detection methods (e.g. Calabresi 1995), of which extensive reviews are provided in Liu *et al.* (2004) and Lu *et al.* (2004). Image statistics-based active contour models (Mason and Davenport 1996; Horritt 1999) have been used by some authors to successfully extract a flood shoreline, for which Mason *et al.* (2007b) have proposed an improvement based on LIDAR DEM constraining.

Classification accuracies of flooded areas (most of the time defined as a ratio of the total area of interest where classification errors are omitted) vary considerably and only in rare cases exceed 90%. Interpretation errors (i.e. dry areas mapped as flooded and vice versa) may arise from a variety of sources: inappropriate image-processing algorithm, altered backscatter characteristics, unsuitable wavelength and/or polarizations, unsuccessful multiplicative noise (i.e. speckle) filtering, remaining geometric distortions, and inaccurate image geocoding. Horritt *et al.* (2001) state that wind roughening and the effects of protruding vegetation, both of which may produce significant pulse returns, complicate the imaging of the water surface. Moreover, due to the corner reflection principle (Rees 2001) in conjunction with its coarse resolution, currently available SAR is unable to extract flooding from urban areas, which for obvious reasons would be desirable when using remote sensing for flood management. Note that recently launched SAR satellites with higher spatial resolution (1–3 m) and carefully chosen incidence angle and wavelength (e.g. TerraSAR-X; Fig. 11.4) may allow reliable flood extraction from urban areas after careful subtraction of radar shadow and layover modelling from LIDAR (Mason *et al.* 2010).

Generally, the magnitude of the deteriorating effects, which determines the choice of an adequate processing technique, is a function of spatial resolution, wavelength, radar look angle and polarization. Henry *et al.* (2006) compared different polarizations for flood mapping purposes and concluded that HH (horizontal transmit–horizontal receive) is most efficient in distinguishing flooded areas.

Water stage retrieval

Direct measurements

Space-borne image-based **direct** measurements have only been obtained from the Shuttle Radar Topography Mission (SRTM) flown in February 2000 (Alsdorf *et al.* 2007). Despite the degraded vertical accuracies over inland water surfaces (up to ± 18.8 m) of SRTM DEMs, LeFavour and Alsdorf (2005) showed that these globally and freely available data may be used to extract surface water elevations and estimate a reliable surface water slope, provided that the river reach is long enough. Kiel *et al.* (2006) assessed the performance of X-band and C-band SRTM DEMs for the Amazon River and a smaller river in Ohio. They concluded that the C-band SRTM DEM gives reliable water elevations for smaller river reach lengths as well. They also state that while SRTM data are viable for hydrological application, limitations such as the along-track antennae offset and the wide look-angle suggest the necessity of a new satellite mission (SWOT – Surface Water Ocean Topography; http://decadal.gsfc.nasa.gov/swot.html) for improved water elevation acquisition.

For changes in water stage retrieval with InSAR technology, the specular reflection of smooth open water causes most of the return signal to be reflected away from the antenna, rendering interferometric retrieval difficult if not impossible. However, for emerging vegetation in inundated floodplains, Alsdorf *et al.* (2000, 2001, and also 2007 for a short review) show that it is possible to obtain reliable interferometric phase signatures of water stage changes (at centimetre scale) in the Amazon floodplain from the double bounced return signal of the repeat-pass L-HH-band Shuttle Imaging Radar (SIR-C). L-band penetrates the vegetation canopy and follows a double bounce path that includes the water and tree trunk surfaces,

Fig. 11.4 A 3-m-resolution TerraSAR-X image of flooding in Tewkesbury, UK, in July 2007 (dark regions are water and radar shadow areas). © DLR (2007).

with both amplitude and phase coherence stronger than surrounding non-flooded terrain, permitting determination of the interferometric phase (Alsdorf *et al.* 2001). Alsdorf (2003) also used these characteristics and found that decreases in water levels were correlated with increased flow-path distances between main channel and floodplain water bodies that could be modelled in a GIS. This correlation function allowed changes in water storage to be mapped over time.

Altimeters (onboard ERS, ENVISAT or JASON mission satellites) emit a radar wave and analyse the return signal. Surface or water height is the difference between the satellite's position in orbit with respect to an arbitrary reference surface and the satellite-to-surface range. Although range accuracies usually lie within 5 to 20 cm for oceans and sea ice (Rees 2001) but typically ~50 cm for rivers (Alsdorf *et al.* 2007), the altimeter footprint is only in the range of 1 to 5 km and seems thus only suitable for rivers or inundated floodplains of large width (Birkett *et al.* 2002; Fig. 11.5). For large lakes accuracies may improve to <5 cm root mean squar (RMS) error (Birkett *et al.* 2002; Alsdorf *et al.* 2007). Another disadvantage of altimetry for water stage retrieval over land is that its success relies primarily on adequate re-tracking of complex contaminated waveforms (Garlick *et al.* 2004).

However, the launch of ICESat in 2003 has made space-borne LIDAR altimetry available for terrestrial water bodies with an elevation precision of a few centimetres (Frappart *et al.* 2006), suitable for detecting river surface slopes along long river reaches or between multiple crossings of a channel. Also the potential to measure water stage beneath vegetation (Harding and Jasinski 2004) could prove interesting for flood monitoring and modelling.

Indirect measurements

Some interesting developments in extracting water levels from remote sensing are those that integrate topographic data (Raclot 2006). Topographic maps with small interval contours and level data may provide an excellent ground truth check for water levels on flood shorelines from aerial photography (Currey 1977) or satellite imagery (Oberstadler *et al.* 1997; Brakenridge *et al.* 1998). LIDAR or photogrammetric DEMs can be intersected with lines from flood deposits on aerial photographs (Lane *et al.* 2003) or high-resolution space-borne imagery from visible bands. Even heights from flooded vegetation may be used (e.g. Horritt *et al.* 2003).

The floodplain can also be segmented into polygons in which water levels are supposed to be horizontal, similar to possible approaches with most 1-D hydrodynamic models. In order to ensure a decreasing water trend with flow direction, extracted water stages are adjusted using an automated algorithm based on hydraulic constraints [see Puech and Raclot (2002) for application to

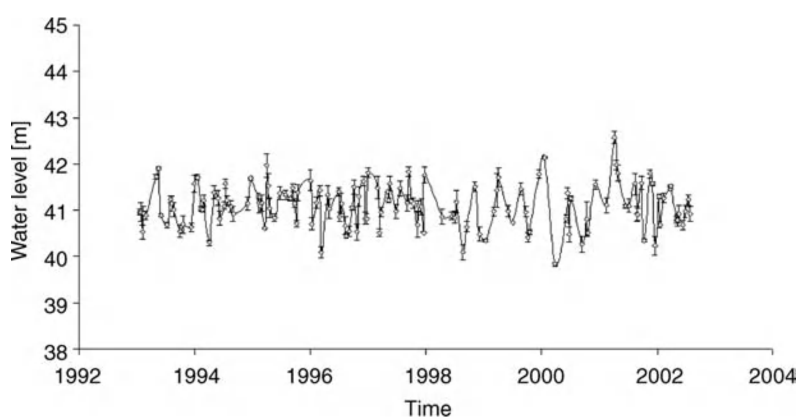

Fig. 11.5 River Danube water level fluctuations from 1993 to 2002. Bars indicate the standard deviation. Data provided by the LEGOS hydroweb (http://www.legos.obs-mip.fr/en/equipes/gohs/resultats/i_hydroweb).

aerial photography; and Hostache *et al.* (2009) for an application to SAR]. Used in conjunction with topographic maps or LIDAR this might lead to vertical RMS accuracies of around 20–30 cm.

Retrieval techniques that combine imagery with LIDAR topography and statistical data analysis have also been suggested. River cross-sections are drawn perpendicular to the main channel, and elevation data are extracted at the SAR flood boundaries, assuming a horizontal water level at each section. A smoothed linear trend of water levels is estimated by using either a moving average filter or spline interpolation (Matgen *et al*. 2007a). A least squares estimation in flow direction is another approach for water surface approximation with respect to localized flow behaviour (Schumann *et al*. 2007a). Additionally, regression modelling allows reliable simulation of stages at any location along the stream centreline (with an RMS accuracy of <20 cm). These data can be used in a GIS to generate a triangular irregular network (TIN) mesh of coherent flood area and stage across the inundated floodplain. As regression modelling, particularly linear modelling, may be undesirable when integrated with more dynamic hydraulic models, multiple water stage data points may be extracted on river cross-sections (Schumann *et al*. 2008a). This allows descriptive statistics (e.g. mean, median or quartiles) to be applied instead of a least squares estimation. The advantage is that levels are now considered varying perpendicular to as well as in the direction of stream flow, with a median accuracy generally better than 50 cm.

Integration with inundation models

Building and understanding model structures

As reported in the first part of this chapter, remotely sensed data are crucial to build models and define boundary conditions. Integration of remote sensing has also been used to understand the difference in behaviour of different model structures. Localized error information, resulting from a comparison of model simulations with spatially distributed SAR-derived water stages, may be used to attribute differences in model behaviour to differences in channel roughness (Schumann *et al*. 2007b). This allows the definition of a model structure that uses additional roughness parameters in order to strike the balance between model complexity and performance at the local level where accurate field observations are available.

Quantifying model performance based on flood area observations

The most common procedure to assess model performance is through an overlay operation of single or multiple simulations of flood inundation models with binary maps from remote sensing based on wet/dry cells in a GIS. For this operation, map outputs from 2-D models can be readily used with a contingency table (also called confusion matrix) that counts the number of correctly and incorrectly flooded/non-flooded cells. One-dimensional models require adequate post-processing of their output to render a binary map comparison possible. There exist a vast number of performance measures based on these categorical data, of which Table 11.3 provides a short summary and recommendation for flood studies (Hunter 2005). However, the trouble with area-based performance measures is that (i) there is not a single best one, and (ii) each one is difficult to interpret, mostly because easily predictable dry cells within the contingency table are misleading.

As an alternative to single area-based performance measures, the modeller may use a comparison of multiple measures or use a fuzzy-rules-based measure, where a simple yes/no (i.e. wet/dry) answer is augmented by a ‘maybe’ relative to the certainty of a given cell being flooded (Pappenberger *et al*. 2007). Such fuzzy membership functions have been used successfully to evaluate multiple model simulations within an uncertainty framework, such as the Monte Carlo-based Generalised Likelihood Uncertainty Estimation (GLUE; Beven and Binley 1992). A membership function reflects the lack of knowledge about the real flood extent as

Table 11.3 Recommendations for various model performance measures (after Hunter 2005)

Name	Remarks	Equation	Recommendation
Bias	Predictions that count (A,B,C[1])	$\frac{A+B}{A+C}$	**Recommended for summarizing aggregate model performance** (i.e. under- or overprediction)
PC	Heavily influenced by the most common category and hence, implicitly domain size	$\frac{A+B}{A+B+C+D}$	**Not recommended for either deterministic or uncertain calibration** The values for *D* are usually orders of magnitude larger than the other categories and may also be trivially easy to predict. Therefore, in many instances, *PC* will provide an overly optimistic assessment of model performance
ROC analysis (F,H)	Artificial minimizing and maximizing of F and H respectively	$H=\frac{A}{A+C}$ $F=\frac{B}{B+D}$	Summarizes two different types of model error (i.e. under- or overprediction) that can occur and is potentially a useful tool for exploring their relative consequences and weighting in any subsequent risk analyses. Therefore, **worthy of further consideration/development**
PSS	Underprediction relative magnitudes of *F* and *H*	$\frac{(AxD)-(CxB)}{(B+D)x(A+C)}$	**Not recommended for either deterministic or uncertain calibration**. Small *F* and large *H* are typical in this type of application and, as such, the measure fails to adequately penalize overprediction. Significant overestimates of the flooded area are therefore only graded slightly poorer than optimal simulations. This also results in the preferential rejection of underpredicting parameter sets during uncertain calibration
$F^{\langle 1\rangle}$	Correct prediction of flooding	$\frac{A}{A+B+C}$	**Recommended for both deterministic and uncertain calibration**. A relatively unbiased measure that simply and equitably discriminates between under- and overprediction. As such, optimal simulations will provide the best compromise between these two undesirable attributes
$F^{\langle 2\rangle}$	Overprediction	$\frac{A-B}{A+B+C}$	**Recommended for deterministic calibration (if underprediction is preferable)**. Explicitly penalizes overprediction but suffers as a result during uncertain calibration. Overpredicting simulations are wrongly retained to offset the bias introduced by the measure and provide an acceptable compromise between inundation map accuracy and precision. The benefits of rejection are reduced accordingly
$F^{\langle 3\rangle}$	Underprediction	$\frac{A-C}{A+B+C}$	**Recommended for deterministic calibration (if overprediction is preferable)**. Here the measure was not tested within the uncertain calibration methodology though for other reaches, events and study objectives, $F^{\langle 3\rangle}$ may provide a useful alternative to $F^{\langle 1\rangle}$ and $F^{\langle 2\rangle}$. It is not sensitive to domain size and appears to favour overprediction similar to PSS

[1]A = area correctly predicted as flooded by model.
B = area predicted as flooded that is actually non-flooded (over-prediction).
C = area predicted as non-flooded that is actually flooded (under-prediction).
D = area correctly predicted as non-flooded.

it allows one to express one's belief in a pixel being flooded and assign a performance value to a simulation that predicts the pixel as flooded accordingly (Matgen *et al.* 2004; Pappenberger *et al.* 2006). Additionally, such an approach may give insights into the effects of different model parameters on acceptability of model performance (Schumann *et al.* 2007b).

Visually illustrating uncertainty in model performance may be a difficult task given the 'fuzziness' of the information content and the complex model parameter interactions when dealing with multiple model simulations. Nevertheless, there have been a few notable attempts to output uncertain flood maps. Romanowicz *et al.* (1996) proposed to derive a 'probability' map by:

$$RCM_j = \frac{\sum_i L_i w_{ij}}{\sum_i L_i} \tag{11.1}$$

in which L_i is the weight for each simulation i, and the simulation results for the j^{th} model element (e.g. computational cell or node) is $w_{ij} = 1$ for wet and $w_{ij} = 0$ for dry. The weight can be based on normalized performance measures, which are derived from maps conditioned on remotely sensed information. RCM_j is the relative confidence measure for each cell j, which expresses a belief that an uncertain prediction is a consistent representation of the system behaviour.

Horritt (2006) addressed the issue by exploiting the spatial nature of floods and computing model precision and accuracy over the model domain. A precise map will contain large areas that are classified as definitely dry or wet, and few areas of probability around 0.5. The precision of the map can therefore be measured by the entropy – defined as C in Horritt (2006). For an accurate uncertainty map, the regions with probability 0.5, for example, will contain equal areas of wet and dry observations. The accuracy can therefore be visualized by the reliability curve, which plots the model probability against the proportions of wet and dry areas in the observations (Fig. 11.6). An accurate model will exhibit a 1:1 relationship, and the deviation from this (e.g. the RMS error) can be used as a measure of the accuracy (Schumann *et al.* 2009).

Model performance based on water levels

Despite the fact that some authors have demonstrated that water depth might constrain the uncertainty in flood inundation models more efficiently than binary patterns (e.g. Werner *et al.* 2005; Mason *et al.* 2009; Schumann *et al.* 2008b), studies that refer to the use of remote-sensing water levels in the model calibration or validation processes are at present very limited (Schumann *et al.* 2007b, 2008b; Hostache *et al.* 2009; Mason *et al.* 2009). Possible reasons for this include the greater data-processing skills involved in water level retrieval and also the lack of precision often associated with indirectly retrieved water levels. Moreover, although directly retrieved water levels may possess the desired accuracy, they often lack the required spatial resolution. In the case of indirect measurements the inaccuracy has largely been the result of a combination of uncertain flood boundary position and DEMs that are inappropriate for the scale of the river reach under study. However, this situation has considerably improved with the availability of LIDAR and recently developed innovative stage retrieval techniques (as described earlier), and this improvement is likely to continue with the newly launched higher-resolution SAR sensors.

Water stages may be used to define additional flood model parameter classes according to different magnitudes of model error (Schumann *et al.* 2007b). This highlights the importance of model evaluation at the local scale. In a similar approach, Mason *et al.* (2009) used the Student's t-test on the error information between SAR-derived waterlines and modelled ones to define model performances with an *a priori* defined uncertainty level. Another useful implementation is to use the uncertainty associated with water levels to set a spatially continuous acceptability interval (Beven 2006) inside which model simulations are required to fall (see Schumann *et al.* 2008b; Hostache *et al.* 2009). This evaluation procedure allows the modeller to gain insights of the model functioning at different spatial scales (Schumann *et al.* 2008b).

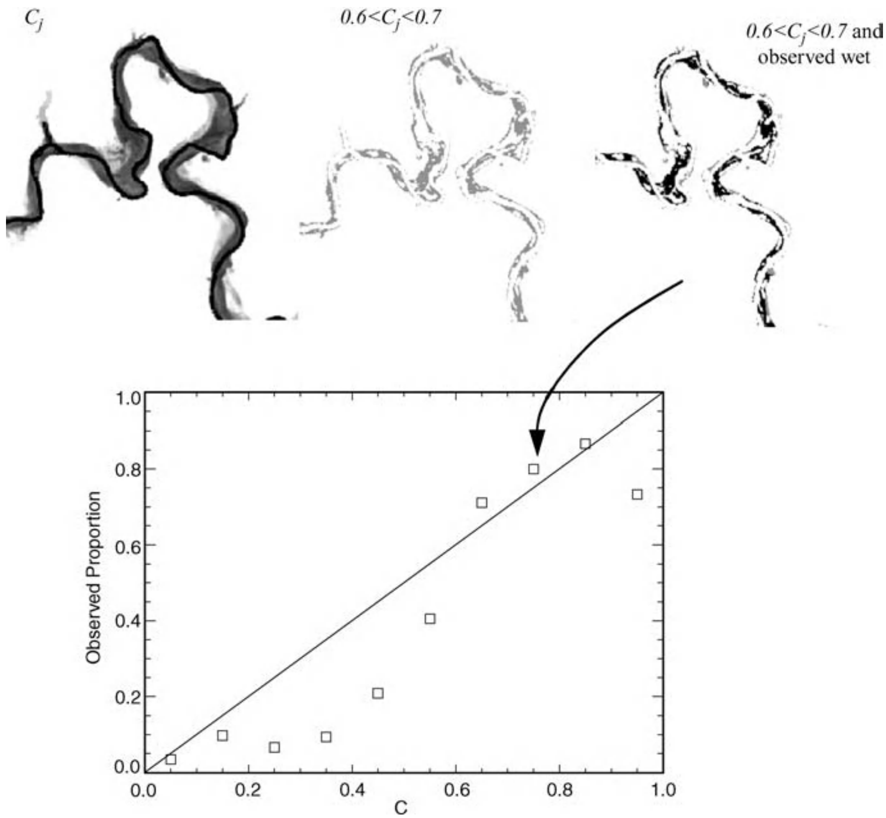

Fig. 11.6 The reliability diagram. The uncertain prediction (top left) is classified into areas of similar Cj (top middle), then the ratio of observed wet/dry cells (top right) calculated. These then make up the reliability plot (bottom), with a reliable model being one with points close to the 1:1 line. After Horritt (2006).

With the launch of new radar satellites and missions (e.g. RADARSAT-2, ALOS, Cosmo-SkyMed, TerraSAR-X and SWOT) with better spatial and radiometric resolutions, the uncertainties of water level estimates will presumably be further reduced, getting closer to the desired centimetre-scale accuracy.

Water stage assimilation into inundation models

Studies in the field of assimilation of remote-sensing data in flood forecasting systems are at present only very few in number, as there are a number of considerable challenges to be faced. Errors in the upstream boundary inflow (Andreadis *et al.* 2007; Matgen *et al.* 2007b) and channel roughness have been considered the only sources of model error, and other likely sources of model uncertainty such as floodplain roughness, channel and floodplain topography, and model structure have been neglected. There is a consensus that a simple reinitialization of models with distributed water stages obtained from remote sensing does not lead to significant improvement because of the dominating effect of the forcing terms on the modelling results (Matgen *et al.* 2007b). By sequentially confronting models with remote-sensing observations it becomes possible to 'diagnose' the latest model set-up and to find out how modelling can be improved.

In this respect, the use of error forecasting models (Andreadis *et al.* 2007; Neal *et al.* 2007) in the context of spatially distributed gauge measurements seems to indicate a promising way forward. A persistent improvement can only be obtained by looking for, and identifying, the reasons that cause disagreement between model results and observations. The objective must be to identify and correct components that are responsible for the discrepancy between modelled and observed variables.

It may be argued that such a 'diagnostic approach' (Gupta *et al.* 2008) ensures efficiency when assimilating remote sensing into flood models. It is reasonable to assume that the difference between the observed and simulated water surfaces might indicate that the model set-up is questionable. An adequate understanding of all the different error sources and interactions is needed to conduct the model development in a meaningful way. Since flood inundation models are calibrated with data of a past flood event, the potential reasons for a mismatch can indeed be numerous: the rating curve used to describe the boundary condition might become erroneous after a given flow magnitude; model parameters (i.e. channel and floodplain roughness values) may vary with time (e.g. due to vegetation growth); important intermediate inflows may have been neglected; or the model structure may become inappropriate with increased inflow (Schumann *et al.* 2009). Put more bluntly, if there is considerable disagreement between observations and model simulations, then both need to be questioned and improvement of the modelling should be envisaged.

Conclusion and Future Research

The foregoing has hopefully illustrated the wide variety of ways in which data are currently being utilized in flood inundation models. Substantial progress can be seen to have been made over the past decade in the development of new data sources, and on the integration of the data they produce into the models. The improvement in urban flood modelling made possible by the availability of high-resolution LIDAR data is but one example. Whilst the progress made to date is significant, much further work remains to exploit fully the data from current sources in the modelling process. In addition, anticipated developments in data sources in the near future mean that the ongoing revolution in the production of data for inundation models is likely to proceed for some time yet. These developments suggest a number of topics for future research to better meet the data requirements of inundation modellers, and these are set out below.

Flood inundation models are only as good as the data used to validate them. There is a need for better model validation datasets, particularly in urban areas. In rural areas, 2-D flood models have been successfully validated using flood extents determined from SAR data, typically ERS and ASAR. However, these have too low a resolution for use in urban areas. This situation should improve in the near future as the number of operational SARs and their spatial resolutions increase. The high-resolution TerraSar-X, RADARSAT-2, ALOS PALSAR and the first two of the COSMO-SkyMed satellites have recently been launched. When the four satellites in the COSMO-SkyMed constellation become operational, a flood revisit time of a few hours should be possible. However, even with resolutions of only a few metres, the side-looking nature of SAR means that substantial areas of ground surface will not be visible due to shadowing and layover caused by buildings, and it will be necessary to correct for these in estimating flood extent (see Fig. 11.4).

The number of operational SARs, coupled with the rise in importance of the Disasters Charter Agency and the advent of the European Space Agency's Heterogeneous Missions Accessibility (HMA) project, also bodes well for the production of SAR image sequences for future flooding events. The Disasters Charter Agency (www.disasterscharter.org) aims to provide a unified system of space data acquisition and delivery to those affected by disasters such as flooding via its member space agencies. The HMA project is establishing harmonized access to Earth Observation data from multiple mission ground segments. This will include a single 'one-stop' mission planning and programming service to place requests for new acquisitions on to partner space agencies' ground segments. The SAR image sequences acquired may image the rising limb of the hydrograph as well as the more commonly imaged falling limb. The availability of image sequences should make possible more data assimilation studies, which may provide more rigorous model validation than using single SAR scenes. If the SAR images can be made available in georegistered form in near real time, they may also become a powerful tool in operational flood risk alleviation scenarios.

Full-waveform LIDAR data need to be processed to produce more realistic topographic data in urban areas. Unlike LIDAR systems recording first and last return, full-waveform systems are able to record the entire backscattered signal of each laser pulse (Chauve *et al.* 2007). Subtle modelling errors may arise due to the limited sampling of the LIDAR waveform that is currently employed (e.g. last return). For example, a wall may divert flood water, but may not be identified in LIDAR data because it is obscured by vegetation, which may subsequently be removed by the filtering process to leave an estimate of ground rather than wall height. Further studies are also required of the fusion of LIDAR data with map and other data in urban areas, and the relative tradeoffs between grid and subgrid representation of urban features.

The improvement of remotely sensed data sources for model parameterization and validation may, in the future, mean that our ability to gauge river flow accurately may become the limiting uncertainty in flood risk modelling. Many gauging stations are located for low-flow monitoring, and perform poorly during high-flow periods. Moreover, obtaining accurate flow velocity data at high flow across a complex floodplain may be difficult and dangerous. Improved flow gauging is therefore likely to be a critical research need in the coming decade, and techniques to achieve this can currently only be described as experimental at best. Boat-mounted Acoustic Doppler Current Profiler (ADCP) technology may provide the most plausible solution here, but it is likely that such technology will need to be used in conjunction with high-resolution flow modelling in order to extrapolate the limited measurements we are likely to obtain to extreme flow conditions. An alternative, using airborne remote sensing, may be to image the flood using airborne SAR with an along-track interferometric capability, allowing measurement of water surface velocity, from which flow rate may be inferred (e.g. Costa *et al.* 2000).

Lastly, networks of low-cost sensors connected using wireless computing and GSM technology (e.g. Neal *et al.* 2007) may also provide an additional source of model validation data to complement that available through remote sensing. The ability to deploy large numbers of sensors may help overcome the spatial resolution of existing ground sensor networks and yield new insights into the hydraulics of flood flows that can be used to develop a new generation of flood inundation models.

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12 Flood Inundation Modelling to Support Flood Risk Management

GARETH PENDER AND SYLVAIN NÉELZ

Introduction: the Role of Modelling in Flood Risk Management

Recent major flood events in Europe and elsewhere (see Chapter 1) linked to growing concerns of the impact of climate change on flood magnitude and frequency have resulted in many governments adopting policies of flood risk management. As discussed previously, this requires a dynamic assessment of risk accounting for both spatial and temporal changes to the flooding system. Such an assessment is not possible without the adoption of a framework within which the various drivers and pressures for change can be systematically evaluated. One such framework, Drivers, Pressures, States, Impacts, Responses (DPSIR), as suggested by Wheater *et al*. (2007), is shown in Figure 12.1.

The techniques discussed in this chapter relate to the 'Modelling of state' (see box highlighted blue in Fig. 12.1) within this framework. In particular this chapter relates to predicting the performance of 'pathways' within the flood system. Pathways are essentially conveyance routes for flood water and may include rivers, estuaries, coasts, pipes and floodplains. The chapter reviews a subset of modelling methodologies available ranging from one-dimensional (1D) methods based on the St-Venant equations through to two-dimensional (2D) methods utilizing the shallow-water equations. Three-dimensional (3D) techniques are considered outside the scope of the current text, as at the present time their application is generally limited to the simulation of local features within the pathway.

The outcome of the models discussed here is an estimate of the impact of flooding upon receptors (Fig. 12.1). In this respect, predictions of inundation extent, depth and velocity coupled to wave celerity are all essential outputs from the models.

This chapter deals exclusively with the technical aspects of model development and implementation, with examples of model application being left until later chapter.

Modelling Methods

Overview

Flood modelling methods currently in use in flood risk management applications can be divided into a number of approaches, presented in Table 12.1, characterized by their dimensional representation of the flood modelling process, or the way they combine approaches of different dimensions. Applications necessary to support the implementation of flood risk management strategies in the UK are mainly covered by the approaches referred to in Table 12.1 as 1D, 1D +, 2D- and 2D methodologies. These are therefore of greatest interest in the present chapter. 3D methods derived from the 3D Reynolds averaged Navier–Stokes equations can be used to predict water levels and 3D velocity fields in river channels and floodplains. However, significant practical challenges remain to be overcome before such models can be routinely applied

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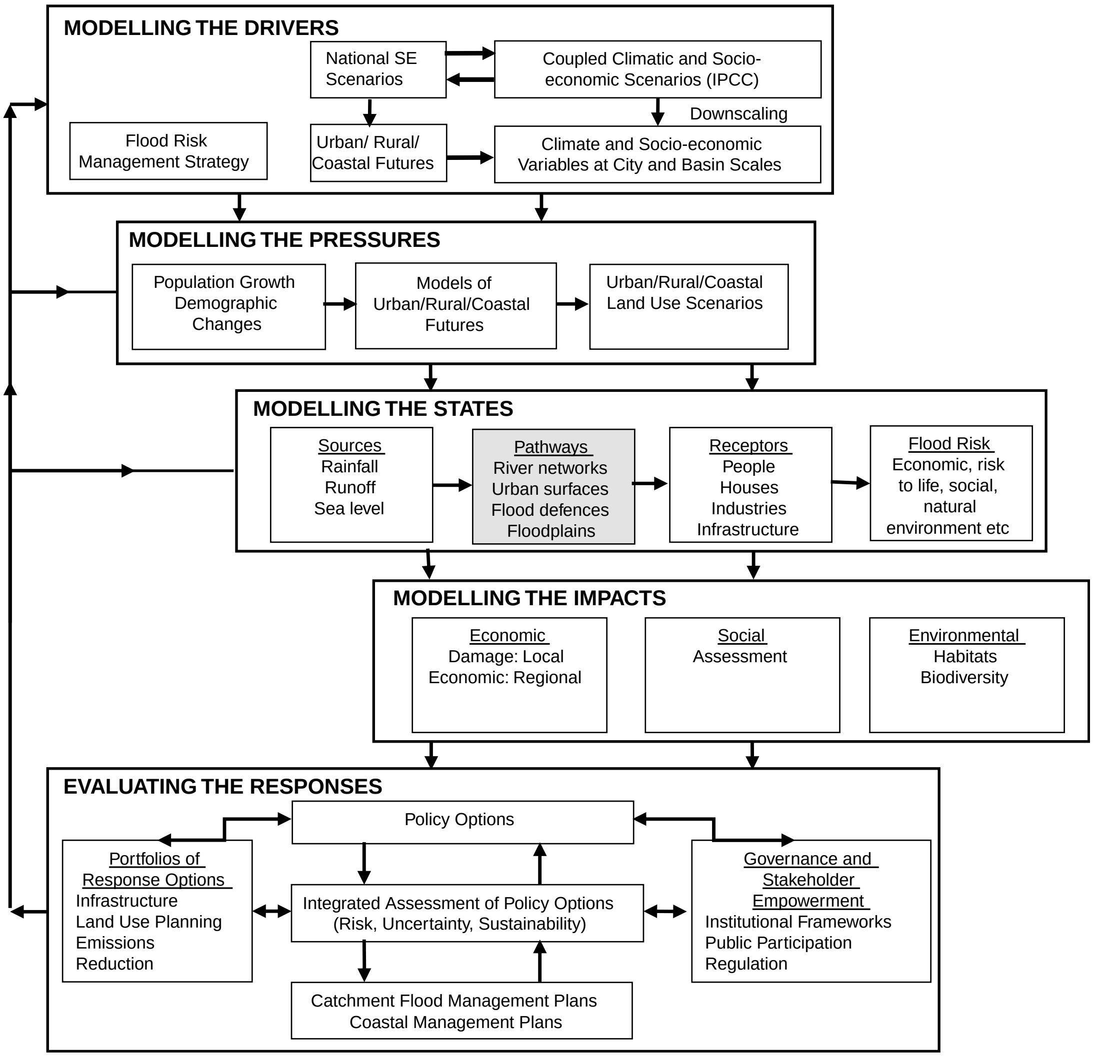

Fig. 12.1 Drivers, Pressures, States, Impacts, Responses-Flood Risk Management (DPSIR-FRM) decision support framework. From Wheater *et al.* (2007). Reproduced with the permission of the Department of Environment, Food and Rural Affairs (Defra).

at the scale necessary to support flood risk management decisions. Table 12.1 provides a summary of the methods and the range of appropriate application for each method.

Most practical problems of floodplain inundation, where horizontal length scales typically exceed flood depths by several orders of magnitude, are best described in 2D. Two-dimensional approaches are almost exclusively based on the two-dimensional shallow water equations. These (also referred to as 2D St-Venant equations, by extension of the use of this terminology; see Hervouet 2007) can be derived by integrating the Reynolds Averaged Navier–Stokes equations over the flow depth. In this integration process a hydrostatic pressure distribution is assumed (see

Table 12.1 Classification of flood inundation models (adapted from Pender 2006)

Method	Description	Application	Typical computation times	Outputs	Example models
1D	Solution of the one-dimensional St-Venant equations	Design-scale modelling, which can be of the order of tens to hundreds of kilometres, depending on catchment size	Minutes	Water depth, cross-section averaged velocity, and discharge at each cross-section Inundation extent if floodplains are part of 1D model, or through horizontal projection of water level	Mike 11 HEC-RAS ISIS InfoWorks RS
1D +	1D plus a storage cell approach to the simulation of floodplain flow	Design-scale modelling, which can be of the order of tens to hundreds of kilometres, depending on catchment size; also has the potential for broad-scale application if used with sparse cross-section data	Minutes	As for 1D models, plus water levels and inundation extent in floodplain storage cells	Mike 11 HEC-RAS ISIS InfoWorks RS
2D-	2D minus the law of conservation of momentum for the floodplain flow	Broad-scale modelling and applications where inertial effects are not important	Hours	Inundation extent Water depths	LISFLOOD-FP JFLOW
2D	Solution of the two-dimensional shallow water equations	Design-scale modelling of the order of tens of kilometres. May have the potential for use in broad-scale modelling if applied with very coarse grids	Hours or days	Inundation extent Water depths Depth-averaged velocities	TUFLOW Mike 21 TELEMAC SOBEK InfoWorks-2D
2D +	2D plus a solution for vertical velocities using continuity only	Predominantly coastal modelling applications where 3D velocity profiles are important. Has also been applied to reach-scale river modelling problems in research projects	Days	Inundation extent Water depths 3D velocities	TELEMAC 3D
3D	Solution of the three-dimensional Reynolds averaged Navier–Stokes equations	Local predictions of three-dimensional velocity fields in main channels and floodplains	Days	Inundation extent Water depths 3D velocities	CFX

Hervouet 2007). A solution to these equations can be obtained from a variety of numerical methods (e.g. finite difference, finite element or finite volume) and utilize different numerical grids (e.g. Cartesian or boundary fitted, structured or unstructured) all of which have advantages and disadvantages in the context of floodplain modelling. Further detailed considerations are provided below.

One-dimensional approaches based on some form of the one-dimensional St-Venant or shallow water equations (Barré de St-Venant 1871) are predominant in river flow studies. Over the years their use has been extended to the modelling of flow in compound channels, i.e. river channels with floodplains. In this case floodplain flow is part of the one-dimensional channel flow, and simulation of inundation is an integral part of the solution of the St-Venant equations. The technique has at least two disadvantages, namely that (i) floodplain flow is assumed to be in one direction parallel to the main channel, which is often not the case, and (ii) the cross-sectional averaged velocity predicted by the St-Venant equations has no physical meaning in a situation where large variations in velocity magnitude exist across the floodplain. The approach has been enhanced in recent years thanks to significant advances in parameterization through the development of the conveyance estimation system (Samuels *et al*. 2002).

Approaches that combine the 1D approach applied to the main channel flow and storage cells to represent floodplains are referred to as '1D+'. These storage cells can cover up to several square kilometres and are defined using a water level/volume relationship. The flow between the 1D channel and the floodplain storage cells is modelled using discharge relationships (such as weir flow equations), which may be used to link storage cells to each other. The water level in each storage cell is then computed using volume conservation. Unlike the 1D approach, the 1D+ approach does not assume that flow is aligned with the river centre line, and therefore may be more appropriate to model floodplains of larger dimensions. However, these models represent wave propagation crudely (water is transferred instantaneously from one end of the storage cell to the other), and calculated inter-cell flows may be significantly in error (because of the lack of appropriate spill discharge equations). The 1D+ approach is also referred to as 'pseudo-2D' (Evans *et al*. 2007) or 'quasi-2D'.

Simplified 2D approaches not based on the full 2D shallow water equations are referred to as '2D-'. This class of models encompasses mainly 2D models based on a simplified version of the 2D shallow water equations where some terms are neglected, resulting in the kinematic and diffusive wave representations (Bradbrook *et al*. 2004; Hunter *et al*. 2007). However, it also includes models relying on square-grid digital elevation models and a simplified 1D representation of the flow between the raster DEM cells (Bates and De Roo 2000). In effect the latter approach is similar to 1D+ approaches but usually with a much finer regular discretization of the physical space. As with the 1D+ approach, momentum is not conserved for the two-dimensional floodplain simulation in 2D- models.

1D, 2D and 3D modelling approaches can also be combined with one another. Many commercially available software packages now offer the possibility to link a 1D river model to 2D floodplain models. This allows modellers to benefit from the advantages of 1D models (computational efficiency, established tradition of 1D modelling), while representing floodplain flows more appropriately in 2D. However, the modelling of the 1D/2D linkage is an area where further research and development is needed, as most approaches in application represent exchange processes rather crudely (see 'Hybrid 1D/2D methods' below). Combined 1D/2D modelling approaches where a 1D sewer system model can be linked to 2D floodplain models are also commercially available.

Finally, some existing models that do not strictly fall in any of the above categories should be mentioned. This is the case of the rapid flood spreading methods (Gouldby *et al*. 2008), which are the subject of research and application in the context of national scale flood risk assessment (for which simulation run times are required to be many orders of magnitude smaller than those obtained using the other approaches listed above). These methods are based on much simpler representations of the physical processes than 2D models and the removal of the time discretization in the computation. In addition, Pender (2006) also refers to a so-called 0D class of inundation modelling methods, which are methods that do not involve any modelling of the physical processes of inundation. One may consider emulation techniques making use of a limited number of training runs of a hydraulic model (see, e.g., Beven *et al.* 2008) to belong to this category. Simple geometric methods consisting in projecting river water levels horizontally over a floodplain can also be termed 0D as far as the modelling of floodplain inundation is concerned. These may be applied to both river and coastal inundation cases.

One-dimensional (1D) flow modelling

The St-Venant equations (Barré de St-Venant 1871) can be expressed as follows:

$$\frac{\partial Q}{\partial x} + \frac{\partial A}{\partial t} = 0 \qquad (12.1)$$

$$\underset{\text{(i)}}{\frac{1}{A}\frac{\partial Q}{\partial t}} + \underset{\text{(ii)}}{\frac{1}{A}\frac{\partial}{\partial x}\left(\frac{Q^2}{A}\right)} + \underset{\text{(iii)}}{g\frac{\partial h}{\partial x}} \underset{\text{(iv)}}{-g(S_0} \underset{\text{(v)}}{-S_f)} = 0 \qquad (12.2)$$

Equation 12.1 is the **continuity** or **mass conservation** equation, and Equation 12.2 is the **momentum conservation** equation. In this, Q is the flow discharge, A is the cross-section surface area, g is the acceleration due to gravity, h is the cross-sectional averaged water depth, S_0 is the bed slope in the longitudinal direction and S_f is the **friction slope** (i.e. the slope of the **energy line**). The various terms in the momentum conservation equation are as follows:

(i) local acceleration term
(ii) advective acceleration term
(iii) pressure term
(iv) bed slope term
(v) friction slope term.

Here the momentum equation is expressed in its **conservative form**. It is possible to substitute UA for Q in Equations 12.1 and 12.2, and rearrange Equation 12.2 to yield the mathematically correct **non-conservative form** of the momentum equation. The use of the non-conservative form may, however, lead to difficulties in its numerical solution (see 'Introduction to numerical methods for inundation modelling' below). Additional terms can be added, such as inflow and loss terms (to the mass conservation equation), and an inflow momentum (to the momentum conservation equation).

Frictional resistance acting on the flow is represented by the friction slope S_f. Several friction slope models exist, namely:

$$S_f = \frac{f}{8gR} U|U| \qquad (12.3)$$

where f is the **Darcy–Weisbach friction factor,** and R is the hydraulic radius ($R = A/P$, where P is the wetted perimeter);

$$S_f = \frac{1}{C^2 R} U|U| \qquad (12.4)$$

where C is the **Chézy coefficient**; and

$$S_f = \frac{n^2}{R^{4/3}} U|U| \qquad (12.5)$$

where n is the **Manning–Gauckler** or **Manning's coefficient** ('Manning's n'). Manning's n is the most commonly applied friction parameter in the UK. The friction slope S_f can be further expressed as follows:

$$S_f = \frac{A^2}{K^2} U|U| \qquad (12.6)$$

where K, the channel **conveyance**, is expressed as:

$$K = \frac{AR^{2/3}}{n} \qquad (12.7)$$

Theoretical assumptions for the St-Venant equations to be valid include: (i) that the bed slope is small; (ii) that the pressure is hydrostatic (streamline curvature is small and vertical accelerations are negligible); and (iii) that the effects of boundary friction and turbulence can be accounted for by representations of channel conveyance derived for steady-state flow.

An alternative form of the momentum conservation equation can be obtained by rearranging Equation 12.2:

$$S_f = S_0 - \frac{\partial h}{\partial x} - \frac{1}{gA} \cdot \frac{\partial}{\partial x}\left(\frac{Q}{A}\right)^2 - \frac{1}{gA} \cdot \frac{\partial Q}{\partial t} \qquad (12.7)$$

It is justified in many applications to neglect the last term in Equation 12.7, yielding the **quasi-steady** form of the momentum equation. In most rivers, the flow is subcritical and all acceleration terms can also be neglected, to yield the so-called **diffusive wave** equation (Julien 2002):

$$S_f = S_0 - \frac{\partial h}{\partial x} \qquad (12.8)$$

A further simplification can be applied by neglecting the pressure term, retaining only:

$$S_f = S_0 \qquad (12.9)$$

which is referred to as the **kinematic wave** equation.

One of the principal strengths of 1D river models is their capability to simulate flows over and through a large range of hydraulic structures such as weirs, gates, sluices, etc. (Evans *et al.* 2007). Recent and ongoing research advances in 1D modelling include enhanced conveyance estimation techniques and afflux estimation techniques. The Conveyance Estimation System (Samuels *et al.* 2002; HR Wallingford 2003) focused particularly on the effects of riverine vegetation, the momentum exchange between the river channel and floodplain flows, and the behaviour of natural shaped channels. It is implemented in a number of commercial packages such as ISIS and InfoWorks-RS. The Afflux Estimation System (Lamb *et al.* 2006) is an improved methodology for the prediction of the increase in water level upstream of a structure (caused by energy losses at high flows through bridges and culverts).

Two-dimensional (2D) flow modelling

The two-dimensional form of the shallow water equations can be expressed as:

$$\frac{\partial U}{\partial t} + \frac{\partial F}{\partial x} + \frac{\partial G}{\partial y} = H \qquad (12.10)$$

where x and y are the two spatial dimensions, and the vectors U, F, G, H are defined as follows:

$$U = \begin{pmatrix} h \\ hu \\ hv \end{pmatrix}; F = \begin{pmatrix} hu \\ g\dfrac{h^2}{2} + hu^2 \\ huv \end{pmatrix};$$
$$G = \begin{pmatrix} hv \\ huv \\ g\dfrac{h^2}{2} + hv^2 \end{pmatrix}; H = \begin{pmatrix} 0 \\ gh(S_{0x} - S_{fx}) \\ gh(S_{0y} - S_{fy}) \end{pmatrix} \qquad (12.11)$$

In this u and v are the depth-averaged velocities in the x and y directions, and S_{ox} and S_{oy} are the bed slopes in the x and y directions. Assuming the use of Manning's n, the friction slopes in the x and y directions can be expressed as:

$$S_{fx} = -\frac{n^2 u\sqrt{u^2+v^2}}{h^{4/3}} \quad \text{and} \quad S_{fy} = -\frac{n^2 v\sqrt{u^2+v^2}}{h^{4/3}} \qquad (12.12)$$

Equation 12.10 reverts the 1D St-Venant equations by ignoring the velocity component and gradients in the y direction and multiplying by the depth-averaged channel width. Equation 12.10 is expressed in conservative form, and similarly with the 1D St-Venant equations, a non-conservative formulation can also be derived.

A number of terms can be added to Equation 12.10 to represent a wider range of physical processes that may contribute to floodplain flow. These include viscosity, the Coriolis effect, wind shear stresses, wall friction stresses, inflow volume and inflow momentum. Viscosity terms F_d and G_d to be added to F and G respectively can be expressed as follows:

$$F_d = \begin{pmatrix} 0 \\ -\varepsilon h \dfrac{\partial u}{\partial x} \\ -\varepsilon h \dfrac{\partial v}{\partial x} \end{pmatrix} \quad \text{and} \quad G_d = \begin{pmatrix} 0 \\ -\varepsilon h \dfrac{\partial u}{\partial y} \\ -\varepsilon h \dfrac{\partial v}{\partial y} \end{pmatrix} \tag{12.13}$$

where the subscript d stands for 'diffusion' as these terms are analogous to a diffusion process. In this, ε is the so-called **viscosity** coefficient, which accounts for the combined effect of (i) kinematic viscosity, (ii) the turbulent eddy viscosity and (iii) the apparent viscosity due to the velocity fluctuations about the vertical average. The contribution of the kinematic viscosity to the value of ε is typically at least an order of magnitude smaller than the turbulent eddy viscosity and for this reason is neglected.

The apparent viscosity due to the velocity fluctuations about the vertical average is recognized as a much more significant contributor to the value of ε (Alcrudo 2004). However, this effect is poorly understood and is therefore also neglected in most applications. The turbulent eddy diffusivity has been the object of significant research (see Rodi 1980), but in the context of flood modelling it is generally not considered an important parameter (Alcrudo 2004). For overland flow conditions it is unlikely that the eddy viscosity will have a major effect on model predictions as friction will dominate. It may, however, have a significant effect upon local high-resolution predictions (Danish Hydraulic Institute 2007a) for flow in and around structures.

Coriolis effects (which account for the effects of the Earth's rotation) are considered negligible in the context of flood inundation studies. Wind shear stresses may result in non-negligible effects on water depths in very large floodplains but their prediction is intimately dependent on the ability to predict wind strength and direction. Wall friction terms are only relevant in very-high-resolution modelling studies and are therefore rarely included.

It is possible to neglect the acceleration terms in the 2D shallow-water equations (the terms involving u and v in U, F and G) to yield the **2D diffusion wave equations** (Bradbrook *et al.* 2004). This is appropriate where the flow is predominantly driven by local water surface slope and momentum effects are less important, as is often the case in the context of UK fluvial floodplains. Such modelling approaches and recent practical applications (e.g. Bradbrook *et al.* 2005) are discussed in Hunter *et al.* (2007).

Introduction to numerical methods for inundation modelling

Classes of numerical methods

Numerical modelling consists of replacing the differential equations such as the shallow water equations by a set of algebraic equations. The process of representing space and time using a finite set of points in the space-time domain and converting the differential equations into algebraic equations is called **discretization**. The many numerical methods in existence can be split into classes depending on the discretization strategy, i.e. the specific approach applied to do this. The great majority of methods used to solve the shallow water equations fall into one of three discretization strategies: Finite Difference, Finite Element, and Finite Volume methods.

Finite difference methods Finite Difference methods rely on Taylor series expansions to express the value taken by a variable (h, u, v, etc.) at a given point, as a function of the values at neighbouring points and of local derivatives of increasing orders. These Taylor series are then combined to yield approximate expressions for the derivatives involved in the shallow water equations, as a function of a finite number of neighbouring point values. The accuracy of the approximations made in doing this can be controlled by the order to which the Taylor series expansions are developed (the order of the so-called **truncation**), which is also linked to the number of neighbouring points involved. In 2D, the implementation of Finite Difference methods is significantly more straightforward on a structured grid (see 'Computational grids' below). This explains to some extent why their popularity is currently in decay in the academic community (Alcrudo 2004), as unstructured grids lend themselves better to the modelling of environmental flows.

Finite element methods In Finite Element methods, the solution space in divided into a number of 2D elements. In each element, the unknown variables are approximated by a linear combination of piecewise linear functions called trial functions. There are as many such functions as there are vertices defining the element, and each takes the value 1 at one vertex and the value 0 at all other vertices. A global function based on this approximation is substituted into the governing partial differential equations. This equation is then integrated with weighting functions and the resulting error is minimized to give coefficients for the trial functions that represent an approximate solution (Wright 2005). A number of methods to do this exist, including the **Galerkin** method (see, e.g., Ottosen and Petersson 1992). Finite Element methods benefit from a rigorous mathematical foundation (Alcrudo 2004), which allows a better understanding of their accuracy (Hervouet 2007); however, the technique has not been used as much as other approaches in commercial software, perhaps because it is less accessible conceptually and produces models that result in large run times. Also, generating meshes can be time consuming when a suitable mesh generation tool is not available (Sauvaget *et al*. 2000).

Finite volume methods In the Finite Volume method, space is divided into so-called finite volumes, which are 2D regions of any geometric shape. The shallow water equations (in conservative form) are integrated over each control volume to yield equations in terms of fluxes through the control volume boundaries. Flux values across a given boundary (calculated using interpolated variables) are used for both control volumes separated by the boundary, resulting in the theoretically perfect mass and momentum conservativeness of the approach (a flux into a finite volume through a boundary is always equal to a flux out of a neighbouring one through the same boundary). In 1D Finite Volume methods are equivalent to Finite Difference methods. Finite Volume methods are increasingly popular and have become the most widely used method in the area of shallow water flow modelling (see, e.g., Sleigh *et al*. 1998; LeVeque, 2002; Caleffi *et al*. 2003; Alcrudo and Mulet 2005; Danish Hydraulic Institute 2007b; Kramer and Stelling 2008). This is explained by their advantages in terms of conservativeness, geometric flexibility and conceptual simplicity (Alcrudo 2004).

Properties of numerical schemes When applying a numerical model from any of the classes above, the local accuracy of the approximation made is controlled by the grid resolution and by the magnitude of local gradients in the flow process. Grid refinement is usually the most obvious way to improve the accuracy of a numerical model. A **convergent** solution is defined as a solution that becomes independent of grid resolution as the grid resolution is increased (Wright 2005).

An important consideration in numerical modelling is the approach used to proceed through the calculation in time. The solution is normally obtained in time-step increments. However, numerical schemes can be divided in two major categories depending on the approach used to discretize the shallow water equations through time. In **implicit schemes** the discretization approach applied to the space gradients involves values at both the previous time step (n) and the new time step ($n + 1$). In **explicit schemes** it involves values from the previous time step only. Implicit schemes are of greater theoretical accuracy. However, the approach also implies that at each new step the solution cannot proceed through the computational grid one node (or finite volume) at a time, and that a system of algebraic equations covering the entire computational domain must be solved. Explicit schemes (which represent the vast majority of newly developed schemes) are simpler to implement. However, they are subject to some form of time step limitation (for stability) analogous to the **Courant–Friedrichs–Lewy**

condition ($u.\Delta t/\Delta x < 1$). Implicit schemes are not subject to such stringent limitations, but time steps are nevertheless limited by considerations of accuracy.

Challenges in numerical modelling

The shallow water equations are non-linear, i.e. they do not satisfy the principle of superposition. One of the implications is that shallow water flows are subject to shock waves, which are to be understood as discontinuous solutions of the shallow water equations (Toro 2001). Shocks on floodplains are mainly encountered in the form of hydraulic jumps, i.e. transitions for supercritical to subcritical flows. These may be caused by local changes in terrain topography (diminution of bottom slope, lateral expansion), or by the effect of bottom friction. An important challenge in the numerical resolution of the shallow water equations is the prediction of the location (celerity) of flood wave fronts and discontinuities (shocks). This is an area where considerable progress has been achieved over the last two decades, mainly through the use of the so-called **shock-capturing** methods. Some of the well-known shock-capturing methods (see Toro 2001) used in inundation modeling include the MacCormack method (Liang *et al.* 2007a), the Lax–Wendroff method, Total-Variation Diminishing (TVD) schemes, Monotonic upstream-centred Schemes for Conservation Laws (MUSCL) based on the Godunov approach, and Essentially Non-Oscillatory (ENO) schemes. Schemes belonging to the class of approximate Riemann solvers (Roe 1981; Toro 1999) are also increasing in popularity.

Two other major areas of ongoing research are related to (i) the treatment of source terms, and (ii) the modelling of wetting and drying. Source terms (H in Equation 12.11) arising from the bed slope dominate in applications to real floodplains, so that the discretization approaches for the flux term and the source terms must ensure an appropriate balance (see, e.g., Garcia-Navarro and Vazquez-Cendon 2000), otherwise spurious movement can be generated by the numerical model even in a body of water at rest. The modelling of **wetting and drying** (the prediction of the boundaries of inundation) is a specific challenge in inundation modelling because flood depths are usually very small along most floodplain inundation boundaries. Model instabilities occur very easily, often due to the fact that friction slope formulae (see, e.g., Equation 12.12) diverge for very small flow depths. A number of approaches have been proposed (see, e.g., Begnudelli and Sanders 2006, or Bates and Horritt 2005 for a comprehensive review of the issue), all of which are a compromise between stability, accuracy and mass conservation.

Computational grids

A **mesh** or **grid** is a collection of points (or **vertices**) where the variables defining the flow condition (velocity, depth or water level) are computed as outlined above under 'Classes of numerical methods'. Closely positioned vertices give a **fine** or **high-resolution** grid, and widely spaced vertices give a **coarse** or **low-resolution** grid. The resolution may also vary in space. The computational efficiency of a numerical model is directly related to the number of equations that need to be solved and therefore to the resolution of the grid.

A **structured** grid is a grid that can be conceptually represented on a rectangular matrix (i.e. the numerical program can effectively make use of rectangular matrices to store the flow variables involved in the computation). Any point in the matrix is physically connected to the four points on either side. A structured grid where the vertices are physically at regular intervals apart is called a structured **square** grid (Fig. 12.2a). A **boundary-fitted** grid is a structured grid that makes use of irregular intervals between vertices (Fig. 12.2b).

An **unstructured** grid is a grid that cannot be represented on a rectangular matrix (Fig. 12.3). The points that constitute such a grid are kept as lists of (x,y,z) coordinates and details on how the points are connected to each other are recorded in a database. The flow variables computed by the model are also stored in the form of lists. The attraction of unstructured grid models lies in the possibility to follow irregular floodplain

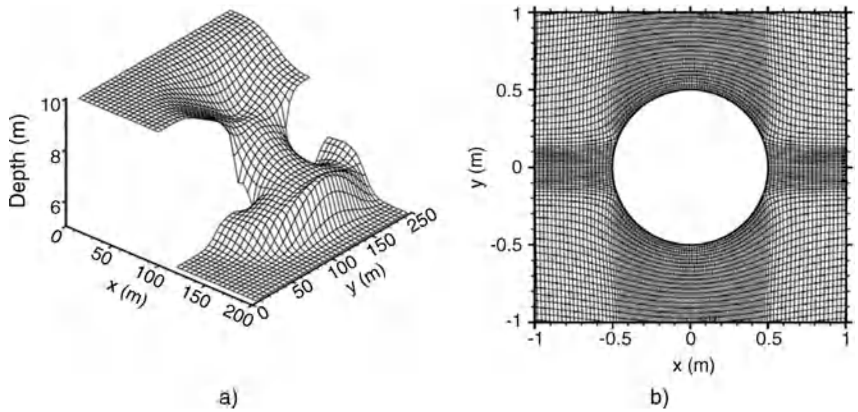

Fig. 12.2 (a) Dam-break simulation on a structured square grid. From Liang *et al*. (2006). (b) Boundary-fitted grid. From Liang *et al*. (2007).

contours, and to apply a non-uniform resolution. It can be refined locally to take into account fine features in the flow, while keeping a low resolution in areas where refinement is not needed, thereby ensuring an optimal use of computer power. However, the finer areas usually dictate that a smaller time step be used, which can increase computation time.

The choice of discretization strategy is linked to the choice of grid type. Finite Difference methods are suited to structured grids only, whereas most Finite Element and Finite Volume methods have been designed with both structured and unstructured grids in mind. Recent advances and current challenges in the area of grid generation are further presented below (see 'Discretization of the physical space').

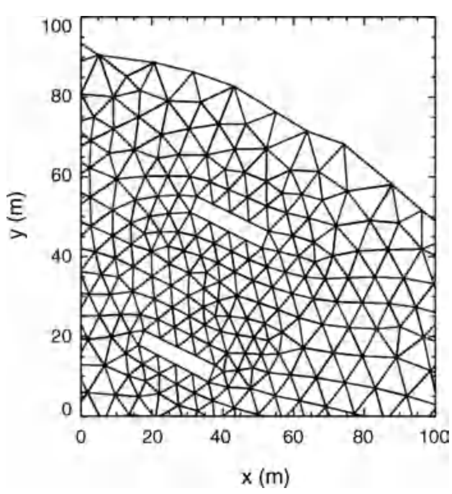

Fig. 12.3 Unstructured mesh. From Horritt *et al*. (2006).

One-dimensional versus two-dimensional

River floodplain modelling is the only context where a comparison of 1D and 2D approaches is relevant. Coastal floodplains can rarely be represented as networks of well-defined channels and therefore 1D floodplain modelling is very rarely appropriate for coastal flooding studies. Also, the theory of open channel flow in the form of 1D St-Venant equations is not applicable to urban flood flows where extreme non-uniformity and spatial variability of flow patterns is common. Flows typically happen in sequences of fast-moving shallow flows (possibly supercritical) and large still ponds, rather than in the form of channels that are well defined over long distances. The significance of storage and recirculation areas that clearly do not fit in a 1D description should not be underestimated. Besides, urban flows rarely happen along routes that are clearly identifiable in advance of building a model and running the simulations (unlike rivers). However, a case where 1D modelling is as close as possible to being appropriate can be found for example in Lhomme *et al*. (2005; deep flooding in a network of well-defined narrow streets).

In river flooding applications, 1D models of rivers with cross-section extending over lateral floodplains are accepted as appropriate for narrow floodplains, typically where their width is not larger than three times the width of the main river channel. An additional condition for such models to be valid is that no embankment, levee or raised ground should separate the floodplain from the main channel. The complicated contribution of the floodplain to conveyance can then be appropriately quantified using recent advances in the estimation of compound channel conveyance (HR Wallingford 2003).

However, 1D river models have limitations that can become significant in many practical applications. The flow is assumed to be unidirectional (generally happening in the direction parallel to the main channel flow), and where this is not true (recirculation areas), conveyance predictions can be severely overestimated. Situations where floodplain flow 'makes its own way' are frequent, but perhaps an even more significant issue is the fact that 1D cross-sections will offer a rather crude representation of floodplain storage capacity in the case of large floodplains.

The use of 1D+ models, where large 'disconnected' floodplains are modelled as storage reservoirs (while narrow floodplains can still be modelled as part of channel cross-sections), allows a better balance between the correct representation of floodplain conveyance and the correct representation of floodplain storage capacity. This latter modelling approach has its own limitations: exchange flows between the river and reservoirs and between the reservoirs are modelled using equations such as broad-crested weir equations (Evans *et al*. 2007), which are not always appropriate. Weir equations adapted for drowned (downstream controlled) flows are also used, but the assumption that water levels are horizontal within each reservoir results in incorrect water level predictions in the vicinity of reservoir boundaries, often causing large errors in the predictions of exchange flows. However, these do not matter if the time duration of the floodplain filling and draining processes is small compared to the duration of the flood. Lastly, the size and location of floodplain storage cells and links between them are user defined and therefore require some *a priori* understanding of flow pathways in the floodplain, which may result in circular reasoning within models.

Two-dimensional modelling of river floodplains can be divided into two important classes of approaches, namely the one where only floodplains are modelled in 2D (as part of a combined 1D/2D model) and the one where floodplain flow and channel flow are modelled as part of the same 2D grid. This latter class of approach is discussed specifically in the final paragraph of this section. The main advantage of 2D modelling (over any other approach for floodplain modelling) is that local variations of velocity and water levels and local changes in flow direction can be represented (Syme 2006). The approach also does not suffer from the limitations of the 1D and 2D$^-$ approaches detailed in the previous paragraphs. It allows in principle a better representation of floodplain conveyance, but a major limitation of combined 1D/2D models for river and floodplain systems is that the exchange processes between the river and the floodplains are still modelled crudely (momentum transfer is not modelled). A major drawback of 2D models is their computational cost (Syme 2006), with a computational run time typically proportional to $1/L^3$, where L is the grid resolution.

Mainly through the use of 2D unstructured grids it is possible to represent a river and adjacent floodplains in a single 2D mesh, Fig. 12.4 (see, e.g., Sauvaget *et al*. 2000; Horritt and Bates 2002). This approach is not particularly common in UK practice, perhaps because there is a long-established tradition of 1D river modelling. Surveyed cross-sections, which are intended primarily for 1D models, exist for a large proportion of rivers in the UK. Numerous existing 1D models have been calibrated using measured data, and 1D Manning's n values are well known for many rivers or river

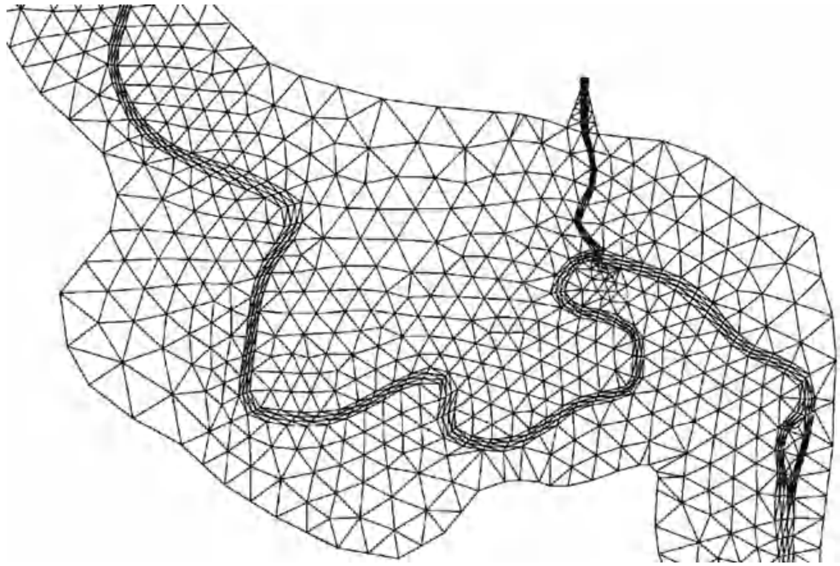

Fig. 12.4 Fully 2D model of the River Severn, UK. From Horritt and Bates (2002).

types. There are therefore benefits in making use of these data and knowledge by continuing to build 1D river models or to use existing ones. In addition, the grid resolution needed to model a river in 2D is significantly finer than what is typically applied on floodplains, resulting in significantly increased computation times. These reasons explain the current enthusiasm for combined 1D/2D modelling for river and floodplain systems (see 'Hybrid 1D/2D methods' below).

Model parameterization and terrain geometry

Parameterization

The friction coefficient (e.g. Manning's n) is the main parameter for which values are required to be set in 2D flood flow modelling. As already mentioned, eddy viscosity is usually considered a secondary parameter and is therefore often ignored. When not ignored, viscosity may be dealt with using a constant viscosity coefficient (see, e.g., Sauvaget *et al.* 2000; Danish Hydraulic Institute 2007a), the Smagorinsky viscosity formulation (Syme 1991; Danish Hydraulic Institute 2007a), or the two-equation k–ε model (Namin *et al.*, 2004). No appropriate methodology exists to calibrate viscosity in flood inundation models, because calibration data at an adequate level of detail do not exist. The viscosity coefficient is sometimes also used to introduce additional artificial viscosity to the flow in order to enhance model stability.

The prediction of flow (velocity and flood wave celerity) is crucially dependent on the friction parameter values adopted in the model. Applications of 1D models benefit from decades of hydrometric data collection, and user experience in model calibration and validation; flood wave propagation (at least in the case of in-bank floods) is now predicted by 1D models with an accuracy that can be considered excellent for many engineering applications. Nevertheless the issue as to whether models should be parameterized using engineering judgment informed by experience, or simply by calibration, or even by an ad hoc combination of both, is still debated in the literature (Beven 2000; Cunge 2003). The parameterization of friction in 2D models benefits to some extent from the knowledge and experience available in 1D modelling, although it should be borne in mind that the formulation of friction is different in 2D models, because (i) bed friction only concerns the interaction of the flow with the river or floodplain bed while in 1D models it concerns the entire wetted perimeter, and (ii) viscosity is explicitly represented in the 2D shallow water equations whereas it is effectively taken into account as part of the friction parameterization in 1D models. Theoretically this should result in lower values (assuming that lower values are used for less rough beds, as is the case with Manning's *n*) of friction in 2D models compared with 1D models (Morvan *et al*. 2008), but in practice the friction parameter is scale dependent and is used to compensate for varying conceptual errors in the model.

The process of parameterizing models of combined river and floodplain systems is made difficult by the problem of **over-parameterization**. Predictions of flood levels by models of compound channels have been shown to be sensitive primarily to the channel friction values used, with the sensitivity to floodplain friction values being much less significant (see, e.g., Pappenberger *et al*. 2005). This reflects the fact that many floodplains act as lateral storage reservoirs where water depths and velocities remain small compared to those in the main channel (it may be argued that this is the case, e.g., in the UK more than in southern regions of Europe where floods can be more extreme). The main consequence is that it is not straightforward to calibrate floodplain friction using measured flood levels (or inundation extent maps; Hunter *et al*. 2005). A more compelling argument for not adopting this approach is the problem of **equifinality** (see, e.g., Aronica *et al*. 1998; Beven 2006), or the non-uniqueness of calibrated parameter values in over-parameterized problems. In the above context it implies that an agreement between model predictions and any observed flood level or inundation extent is achievable by calibrating channel friction values only. In the distinct context where floodplain flow results from some form of failure of flood defences (and continues until flood waters recede or until the floodplain has been completely filled), correct model predictions are then likely to depend on the correct prediction of flood discharge (flowing through or over the failed defence) much more than on the floodplain friction parameterization. The possibility to calibrate floodplain friction in such circumstances using real event data has not been exploited to the present date. It could only exist if (i) the inflow was accurately measured, and (ii) appropriate hydrometric data were collected on the floodplain, i.e. during the transient phase of the inundation process (not after the flood has settled on the floodplain). It can be concluded that inundation extent and floodplain water level measurements alone cannot usually be used to calibrate 2D floodplain models in the same way as river levels are used to calibrate 1D river models (Hunter *et al*. 2005; Werner *et al*. 2005; Néelz *et al*. 2006). In the same way as calibrating 1D models usually involves tuning friction parameter values to yield an optimal match between predicted and measured water level hydrographs, an appropriate approach for 2D floodplain models should at the very least concentrate on the prediction of features of the flow that depend primarily on the processes modelled by the 2D solver (perhaps velocities), and the flow conditions at the boundaries of the floodplain should be known accurately as part of the calibration data. An additional difficulty if measured velocities are to be used as calibration data is that these must be measured in a form that is consistent with what models predict, i.e. depth-averaged velocities.

The above paragraphs focus either on the modelling of floodplains only or on the modelling of floodplains in 2D as part of combined 1D/2D models (see 'Hybrid 1D/2D methods' below). Approaches to calibration for fully-2D models, where floodplains as well as river channels are modelled in 2D, is somewhat different (see, e.g., Sauvaget *et al*. 2000). However, issues of over-parameterization and equifinality affect fully-2D models in a similar manner.

Elaborate approaches to floodplain friction parameterization that do not involve calibration have been suggested in recent years. These are detailed under 'Representation of surface roughness and energy loss' below.

Terrain geometry

The availability of a so-called Digital Elevation Model (DEM) or Digital Terrain Model (DTM) is an important prerequisite in 2D computer modelling of inundation flow (conventions on the use of such terminology are not consistent, but it is often accepted that DEM refers to a representation of the Earth's surface that may or may not include above-ground features such as buildings and other structures, whereas DTM refers to a representation of the 'bare' Earth). In recent years, the generation of DEM and DTM has benefited from very significant advances in the area of remote sensing, involving the automated, broad-area mapping of topography from satellite and, more importantly, airborne platforms. Three techniques that currently show reasonable potential for flood modelling are aerial stereo-photogrammetry, airborne laser altimetry (or LiDAR: light detection and ranging) and airborne synthetic aperture radar interferometry (Asselman *et al.* 2009). LiDAR in particular has attracted much recent attention in the hydraulic modelling literature (Bates *et al.* 2003; French 2003; Smith *et al.* 2005). A major LiDAR data collection programme is underway in the UK, where so far more than 20% of the land surface area of England and Wales has been surveyed. In the UK, helicopter-based LiDAR survey is also beginning to be used to monitor details (~0.2-m spatial resolution) of critical topographic features such as flood defences and embankments. LiDAR systems operate by emitting pulses of laser energy at very high frequency (~5–100 kHz) and measuring the time taken for these to be returned from the surface to the sensor. Global Positioning System (GPS) data and an onboard inertial navigation system are used to determine the location of the plane in space and hence the surface elevation. As the laser pulse travels to the surface it spreads out to give a footprint of $\sim 0.1\,\mathrm{m}^2$ for a typical operating altitude of ~800 m. On striking a vegetated surface, part of the laser energy is returned from the top of the canopy and part penetrates to the ground. Hence, an energy source emitted as a pulse is returned as a waveform, with the first point on the waveform representing the top of the canopy and the last point representing the ground surface. The last returns can then be used to generate a high-resolution 'vegetation-free' DEM, while other returns provide information on the vegetation cover. Buildings will normally be identified by last returns. They can either be kept as part of a DEM, or automatically extracted using specific algorithms (Zhang *et al.* 2003) to produce a DTM.

The typical vertical accuracy of LiDAR data has now reached values below 0.1 m, and recent technical advances in the area of vehicle-mounted LiDAR suggest that LiDAR data accuracy is not a limiting factor for most inundation modelling applications. However, high-accuracy high-resolution ($\ll$ 1-m) LiDAR may not be appropriate for many practical flood modelling applications. It may require excessive computer memory and processing power, but also it is likely that many inundation modelling studies will continue to be carried out at resolutions much coarser than the resolution of DEMs. Most of the information contained in a LiDAR DEM can therefore be redundant when used in inundation models, including details of critical significance to inundation modelling (walls, kerbs, fences, ridges, etc.). This has motivated recent and ongoing research into the development of data-processing techniques that identify such linear features from raw LiDAR data, and generate so-called **breaklines** (Fig. 12.5),

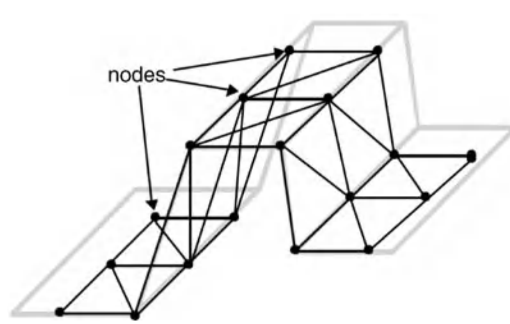

Fig. 12.5 Representation of a dyke in an unstructured model using breaklines. Adapted from Sauvaget *et al.* (2000).

to be used in association with DEM grids in computer models (Romano 2004; Brzank *et al.* 2005; Liu 2008).

Recent, Ongoing and Future Research Challenges

Hybrid 1D/2D methods

Although the ability to link 1D and 2D models has existed in theory for several decades, is has only recently been implemented in commercial software packages (Syme 1991; Evans *et al.*, 2007). It is now becoming increasingly popular because it allows modellers to exploit the respective advantages offered by 1D and 2D models (see above). Possible applications of 1D/2D linking are:

• within a channel that one wishes to model partly in 1D and partly in 2D;
• between a 1D drainage network model and a 2D surface flood model;
• between a 1D river model and a 2D floodplain model;
• within a mainly 2D model where, for example, culverts are modelled in 1D, linking 2D cells to each other.

For example modellers can take advantage of the established tradition of 1D river modelling while at the same time modelling floodplains in two dimensions. It also results in significant computational savings over fully 2D approaches where extreme refinement is required to correctly represent the river channel geometry.

Several numerical techniques exist to date to link 1D and 2D models. The most widely used technique for 1D river and 2D floodplain linking is the **lateral** link (Fig. 12.6), where the exchange flows are based on water level differences and typically modelled using broad-crested weir equations or depth-discharge curves (Lin *et al.* 2006; Danish Hydraulic Institute 2007a; Evans *et al.* 2007; Liang *et al.* 2007b; BMT-WBM 2008). A limitation of the approach is that the complicated momentum exchange processes that characterize the river–floodplain boundary are not modelled. These processes intimately depend on complex 3D flow patterns occurring in the river (Morvan *et al.* 2002), which by definition are not resolved in a 1D river model. Progress towards improved model representation is reported in Liang *et al.* (2007b).

One may also use a **longitudinal** link (Fig. 12.6) to model a watercourse partly in 1D (upstream) and partly in 2D (downstream), or to connect the downstream extremity of a 1D model to a 2D grid (Danish Hydraulic Institute 2007a; Evans *et al.* 2007; Liang *et al.* 2007b). In this approach the flow from the 1D model enters the 2D model as a 'source', and the water level in the 2D model at the junction is used as a downstream boundary condition in the 1D model. Some combined 1D/2D models also offer the possibility to use 1D components to represent pipes or culverts

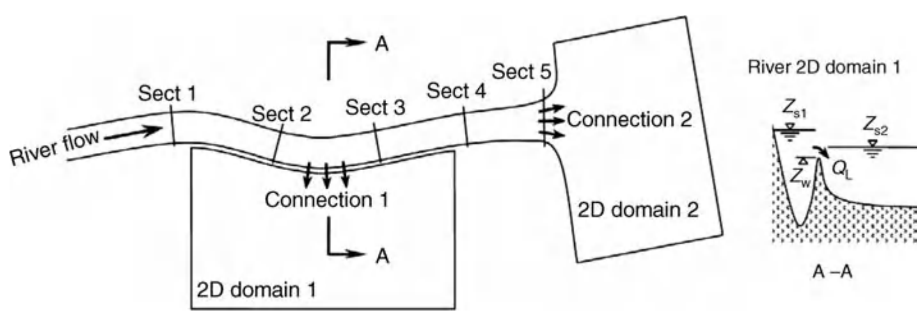

Fig. 12.6 Schematic illustration of the lateral ("Connection 1") and longitudinal ("Connection 2") connection modes. In a lateral link the discharge Q_L is calculated usually as a function of the upstream and downstream water elevations and crest elevation (respectively Z_{S1}, Z_{S1} and Z_W). From Liang *et al.* (2007b).

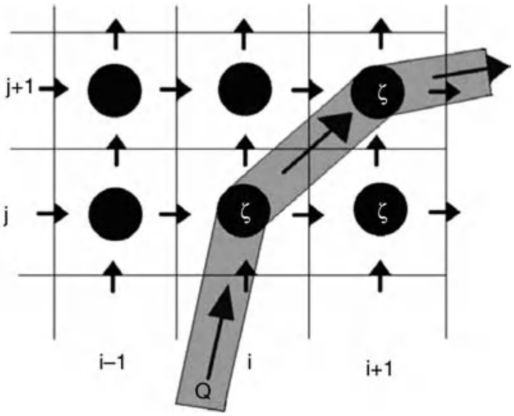

Fig. 12.7 1D-2D flood modelling, as can be used for example in SOBEK. Water exchange occurs in the vertical direction at bankfull level between the river modelled in 1D (discharge Q) and the flood (modelled in 2D). From Stelling and Verwey (2005).

linking nodes within a 2D model, and combined 1D/2D modelling approaches where a 1D sewer system model can be linked to a 2D floodplain model are also commercially available (see, e.g., Rungø and Olesen 2003; Danish Hydraulic Institute 2007a).

Finally the approach consisting in coupling a 1D river model and a 2D floodplain model using a **vertical link** (Fig. 12.7) should be mentioned (Verwey 2001; Bates *et al.* 2005; Stelling and Verwey 2005). This consists in representing the floodplain using an uninterrupted 2D grid overlaying the 1D river model. The 1D model operates on its own until the river reaches bank-full level, at which point the water above this level is transferred to the 2D model.

Representation of surface roughness and energy loss

In 1D modelling representation of surface roughness and other energy losses is normally achieved through a calibration and validation cycle, often with reference to standard hydraulics texts to inform first estimates of the necessary parameters. In the UK, the publication of the Conveyance Estimation System (CES: see www.riverconveyance.net) has advanced the selection of the necessary parameters.

Above we introduced difficulties that inherently hinder the process of calibration of floodplain inundation models, caused either by the non-existence of appropriate calibration data, or by the problem of equifinality, which inevitably prevails in over-parameterized models. These limitations have in recent years motivated the development of approaches to parameterize spatially varying floodplain friction that do not involve calibration. These approaches make use of the wealth of information provided by remote-sensing data such as LiDAR (see above), from which spatially distributed details on vegetation thickness and density can be extracted (Asselman *et al.* 2002; Cobby *et al.* 2003; Mason *et al.* 2003, 2007; Davenport *et al.* 2004). However, it should be kept in mind that output variables such as inundation extent and point water levels may not be sensitive to distributed friction values on river floodplains to any discernible extent, as demonstrated by Werner *et al.* (2005). This suggests that a methodology for floodplain friction parameterization at a coarse scale may be more appropriate than the use of such technologies if output parameters that usually have low spatial variability such as water levels are of interest. In applications where detailed predictions of flow patterns (including velocities) are sought, then spatially distributed friction values have more relevance. Other types of datasets such as high-resolution land-use maps (Mason *et al.* 2007) may help in setting friction parameter values. It should be added that even if friction phenomena on different surfaces (roads, etc.) and through different types of vegetation were adequately understood and modelled, there could remain the issue of modelling very localized processes involving head losses such as those caused by hedges and fences. Again, these may be better taken into account in a model parameterized at a coarse scale where they would effectively be treated as subgrid processes in the same way as bottom friction. Considerable research aiming to design such parameterization techniques for coarse grid models is needed.

Simulation time and the need for rapid methods

In recent decades, efficiency in numerical modelling has been governed primarily by the exponential increase in computer power. The number of transistors that can be placed inexpensively on an integrated circuit has doubled approximately every two years ('Moore's law', Moore 1965), and this trend is continuing to be observed towards 2010. Other factors include progress in the design of efficient numerical methods, although this rarely leads to step changes in computation efficiency, primarily because the Courant–Friedrichs–Lewy–condition imposes a limit on the size of usable time steps in explicit numerical schemes. A similar, although less stringent, limitation affects implicit schemes, in which the use of excessively large time steps leads to unphysical oscillatory solutions.

Recent research in the area of parallel processing should also be mentioned. This consists in using simultaneously a cluster of computers or a cluster of processors within a computer (Hervouet 2007). The challenge is to divide numerical algorithms into subalgorithms that are as independent as possible from each other and between which the amount of information that is transferred is limited. This has been implemented in research codes (Kramer and Stelling 2006; Hervouet 2007; Wright and Villanueva 2008) but commercial applications do not yet exist.

However, the demand on computational efficiency imposed by the need to simulate inundation modelling scenarios as part of risk-based methods (Hall *et al.* 2003; Gouldby *et al.* 2008) makes any advances permitted by the above or by the use of unstructured or coarse meshes insufficient. Flooding can arise from different sources – extreme tidal surges or fluvial flows, for example – occurring on their own or in combination with each other. For strategic planning purposes, timescales of a century or more are under consideration, and uncertainties due to climate change must be taken into account. Defences designed to mitigate flood risk can fail, and failing processes are poorly understood, while the probability of failure changes with time as a result of natural deterioration and maintenance operations. Probabilistic risk-based methods, where risk is understood as combining probability and consequence, therefore need to consider a number of inundation scenarios that can reach thousands to hundreds of thousands. Traditional mesh-based numerical techniques are inapplicable, particularly when such risk-based methods are applied to countrywide scales. This has recently motivated research into so-called 'rapid flood spreading' methods (Gouldby *et al.* 2008), which in their initial form simply consist in pre-processing Digital Elevation Models to separate floodplains into 'impact zones', and establishing a volume-spreading rule based on storage capacity within these impact zones and ground elevations at the communication points. For each scenario of a risk computation the 'rapid flood spreading' calculation succeeds in achieving run times several orders of magnitude shorter than hydrodynamic time-step-based simulations. In their present state these methods do not take into account resistance and momentum effects that may affect the final state of inundation, and do not predict any time-varying flood depth or velocity outputs, which may eventually become desirable to include in risk computations.

Discretization of the physical space

One of the challenges of flood inundation modelling arises from the extreme non-uniformity in the physical dimensions of the processes of interest. European floodplains have typical dimensions ranging from a few dozens of metres to a few dozens of kilometres. Within these, natural landforms and made-made structures such as levees and embankments, typically a few metres in transverse dimensions, may strongly affect flow routes. At the local scale, and primarily in the urban environment, walls and other structures often have a critical impact, and hedges and fences also result in large head losses. For consistency, appropriate approaches should be applied to model these obstacles if attention is otherwise paid to the modelling of surface roughness.

Unstructured grid-models are seen by many as the most promising way forward (Namin *et al.* 2004; Begnudelli and Sanders 2006; Hervouet 2007), because of their potential for non-uniform grid resolution, allowing refinement only where needed and thereby saving computational effort. However, automatic grid generation techniques for unstructured grids are at an early stage of development for applications in flood flow modelling. Most commercial unstructured-grid models still require time-consuming human intervention (Sauvaget *et al.* 2000). However, significant advances in the field (see, e.g., Shewchuk 1996; Owen and Shephard 2003) are beginning to be applied (e.g. Begnudelli and Sanders 2006), including in some commercially available software (e.g. Infoworks-RS 2D; see Gutierrez-Andres *et al.* 2008). 'Smart' grid generation techniques that are specifically designed for floodplain flow modelling, and that integrate physical features of the floodplains digitized in the form of breaklines (see Fig. 12.5) or building outline polygons are being implemented. It is also worth mentioning research algorithms that make mesh resolution locally dependent on vegetation features (e.g. Cobby *et al.* 2003). Such advances are, however, still to be used in engineering practice.

Advantages of structured grids over unstructured grids include algorithmic simplicity. The boundary-fitted grid approach (Fig. 12.2b) (Liang *et al.* 2007a) combines this with the advantage of geometric flexibility, but grid generation for applications to highly irregular geometries is difficult. The alternative of using square-structured grids is still overwhelmingly popular because of its simplicity. Therefore, recent research efforts have concentrated on the development of methods that treat variability in the physical domain and in the topography at the sub-cell level. For example, the Cartesian cut cell method uses a background Cartesian grid for the majority of the flow domain, with special treatments being applied to cells that are cut by solid bodies. The development of the method is described in Ingram *et al.* (2003), and a recent application to inundation modelling is proposed in Morris *et al.* (2006). A different approach relying on the so-called **quadtree** grid generation methodology (Fig. 12.8) has been promoted in recent years (Liang *et al.* 2008; Liang and Borthwick 2009). Quadtree grids are structured square grids that can be locally refined according to criteria associated, for example, to transient hydrodynamic properties of the flow or to spatial topographic variability. Their main feature is that local refinement is only carried out through a recursive spatial decomposition process where a square cell can only be subdivided into four smaller and also square cells. This allows the use of

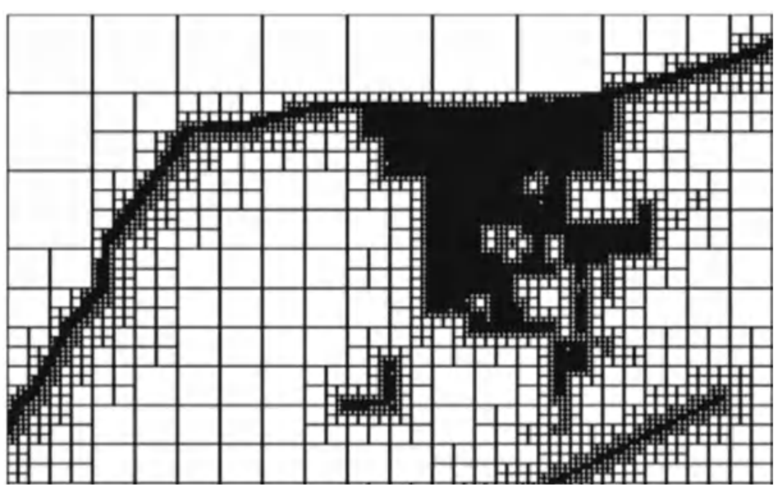

Fig. 12.8 Quadtree grid modelling. From Liang *et al.* (2008).

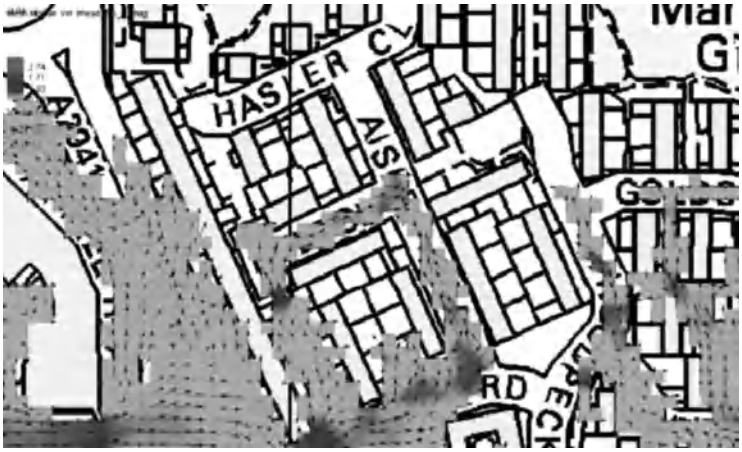

Fig. 12.9 Modelling of urban inundation at high resolution. (See the colour version of this figure in Colour Plate section.).

a simple cell indexing approach in the numerical algorithm, retaining a lot of the algorithmic simplicity of structured grid models while allowing local refinement.

Finally, it must be noted that recent research has investigated the sensitivity of inundation model results to resolution (Horritt *et al*. 2006; Fewtrell *et al*. 2008; Schubert *et al*. 2008). While it can be accepted that refinement normally leads to more accurate solutions, the benefits to be gained from the use of finer grids are far from self-evident because of the the loss in computational efficiency involved.

Risk-based methods (Hall *et al*. 2003) rely on the ability to simulate large numbers of scenarios, taking into account uncertainties in the drivers of flooding. In practical applications these uncertainties are such that performing low-resolution simulations may allow many more of them to be run, and eventually a better estimate of risk (and associated uncertainties) to be computed. In this respect there is an essential difference between model output variables such as water level and velocity. The former is often much more smoothly varying in space than the latter, which may show very steep local gradients particularly in the urban area. The prediction of flood velocity is therefore at the present time somewhat incompatible with risk and uncertainty computations (Néelz and Pender 2008).

The challenges of urban inundation modelling

The limitations of the approach consisting in modelling streets as 1D channels as part of a 1D computational model have already been introduced (see 'Model parameterization and terrain geometry' above; see also Lhomme *et al*. 2005), and most modellers favour two-dimensional approaches. These can be divided into two classes: the high-resolution alternative where flow pathways along streets and between buildings are represented, and the low-resolution option, which often imposes itself in practical applications on grounds of computational affordability, where such flow pathways and their overall effects on the flow field must be somehow modelled as a subgrid-scale physical process. No clear threshold value of the grid resolution allows distinguishing between the two, but it is usually the case that a grid finer than ~5 m will allow most narrow streets and gaps between buildings to be resolved, although this may sometimes have to be refined further, for example in many historical city centres (Haider *et al*. 2003), or in situations involving highly transient flows (Soares-Frazao and Zech 2008). A number of difficulties arise when setting up a high-resolution flood model of an urban environment. First, spatially varying roughness values must be set to account for various surface types (tarmac, grass, any tall vegetation, gardens).

This is facilitated by modern topographic data; however, the decision on the choice of value to apply to each category remains somewhat subjective. For high depth to grid size ratios, the lack of appropriate theory to assist in setting appropriate eddy viscosity values may become a significant issue (Calenda *et al*. 2003; Liang *et al*. 2006). Finally, localized topographic features of small dimensions such as walls and fences have a significant impact on local flow patterns, and data at the appropriate resolution are seldom available.

The modelling of buildings in high resolution is a challenge per se, primarily because real buildings affect flood flows in different ways, which are often unpredictable. Some buildings with wide openings or with openings forced by the strength of the flood may become flooded with little or no time delay, while others may remain dry at least initially. Hydraulic storage space within a building cannot be inferred from maps or aerial images, as some buildings may be elevated in relation to the surrounding ground, while others may have basements. Several approaches are available to include buildings in high-resolution models: the use of porosity (Hervouet 2007), high roughness (Néelz and Pender 2008), or methods where one of the building walls is artificially opened (Syme 2008; Schubert *et al*. 2008).

Practical application of high-resolution regular square grids faces the major difficulty of computational inefficiency. A 5-m model of an area of several square kilometres can typically take hours to run on computer hardware typical of that available to engineers at the end of the 2000 decade, using any of the most widely used software package solving the full shallow water equations. With computational times inversely proportional to the cube of the grid size (as a change in grid resolution usually also implies a change in allowable time step), the only practical approach often consists of using a coarse grid, particularly in large-scale studies. The use of unstructured grid models can somewhat alleviate the issue, but time steps are still to some extent governed by the size of the smallest elements (Calenda *et al*. 2003). There is therefore continued pressure on engineers and researchers to design subgrid-scale methodologies applicable to the urban environment. Various approaches have been suggested, such as the use of a porosity term (e.g. Hervouet 2007), the application of a spatially varying roughness coefficient (Néelz and Pender 2007), use of a coverage ratio and conveyance reduction factor (Chen *et al*. 2008), extraction of subgrid-scale connectivity (Yu and Lane 2006) and the above mentioned cut-cell approach (Morris *et al*. 2006). Future developments in this area will be expected to appropriately represent building blockage effects (Lane and Yu 2008; Sanders *et al*. 2008), but also overcome any issue of over-parameterization.

Conclusions

A vast array of modelling methodologies now exist to predict pathway performance within the flooding system. 1D techniques remain appropriate for situations where a clearly defined one-dimensional pathway exists, such as rivers and pipe systems. In situations where no such clearly defined single pathway exists, recourse must be made to 2D methods. Here a choice is required between those techniques employing the full shallow water equations and those based on simplified equations, such as diffusive or kinematic wave equations. The consequences of this choice, when models are applied to floodplains with complex geometries, are presently unclear; however, ongoing research (Environment Agency, 2009) will soon clarify such choices.

Research is continuing to deliver improvements in 2D modelling methods. In particular in the areas of trans-critical flow simulation, hybrid methods (linking 1D and 2D models) and discretization techniques. Additionally, the systems approach required by flood risk management and the uncertainties associated with predicting pressures and flood sources dictate a need for multiple simulations to facilitate a probabilistic approach to uncertainty analysis. There is therefore a pressing need for faster model predictions through either the development of accurate model emulation techniques or the use of parallel processing to speed up 2D modelling methods. Research is ongoing in each of these areas.

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13 Integrated Urban Flood Modelling

ADRIAN J. SAUL, SLOBODAN DJORDJEVIĆ, ČEDO MAKSIMOVIĆ AND JOHN BLANKSBY

Introduction

Recent urban floods have highlighted the need to better understand the performance of our natural and man-made drainage systems to reduce the risk of urban flooding and to better protect the environment. Such future systems need to be sustainable from a technical, environmental, social and economic viewpoint. Delivery of such sustainable systems is a function of changes in several key drivers, for example climate change, population growth, carbon and water footprints, changing customer behaviour and perceptions, and the need to meet new legislation. To meet these challenges the UK government stimulated the Foresight review (Evans *et al*. 2004, 2008), and this clearly identified that the impact of climate change and increased urbanization will see significant increases in the occurrence of urban floods. In response, the 'Water Strategy for England – Future Water' (Defra 2008) highlights the UK government's long-term vision for water and the framework for water management in England, with a need to forecast, prevent and better manage urban floods. This, together with the Pitt Review (Pitt 2008), which examined the causes of the floods that occurred in the UK in the summer of 2007, with 92 recommendations to improve all aspects of urban flooding from forecasting to emergency planning and response, have provided the UK government and UK water industry with an outstanding opportunity to make a paradigm shift in the way in which flood and surface water is managed. This shift is a primary driver in the Urban Flood Management components of the Flood Risk Management Research Consortium (FRMRC) Phase 1, which comprise the topic of this chapter.

Legislation and stakeholder engagement

The UK has to meet both UK and European Union (EU) regulation, and in respect of surface water management, there are many key stakeholders with a responsibility for urban floods. As shown in Figure 13.1, these include the Department for Environment Food and Rural Affairs (Defra), the Environment Agency (EA), local and planning authorities, the Highways Agency, the water service providers, internal drainage boards, British Waterways and landowners. Therefore the approach has to be integrated and involve all stakeholders. Future Water (Defra 2008) sets out a strategy for the development of Surface Water Management Plans (SWMPs) for the future integrated management of urban floods with guidance published by Defra (2009). This guidance builds on the outputs of the Defra Making Space for Water pilot projects (Defra 2008; Gill 2008), and provides a framework to guide local partnerships (EA, local government, water companies and other stakeholders) to take the necessary steps to prepare for an integrated flood risk assessment, to complete the assessment, to identify remedial measures and options and their cost benefit, and subsequently to implement and review the selected strategies to reduce urban flood risk. This guidance highlights the need for new models for a better understanding

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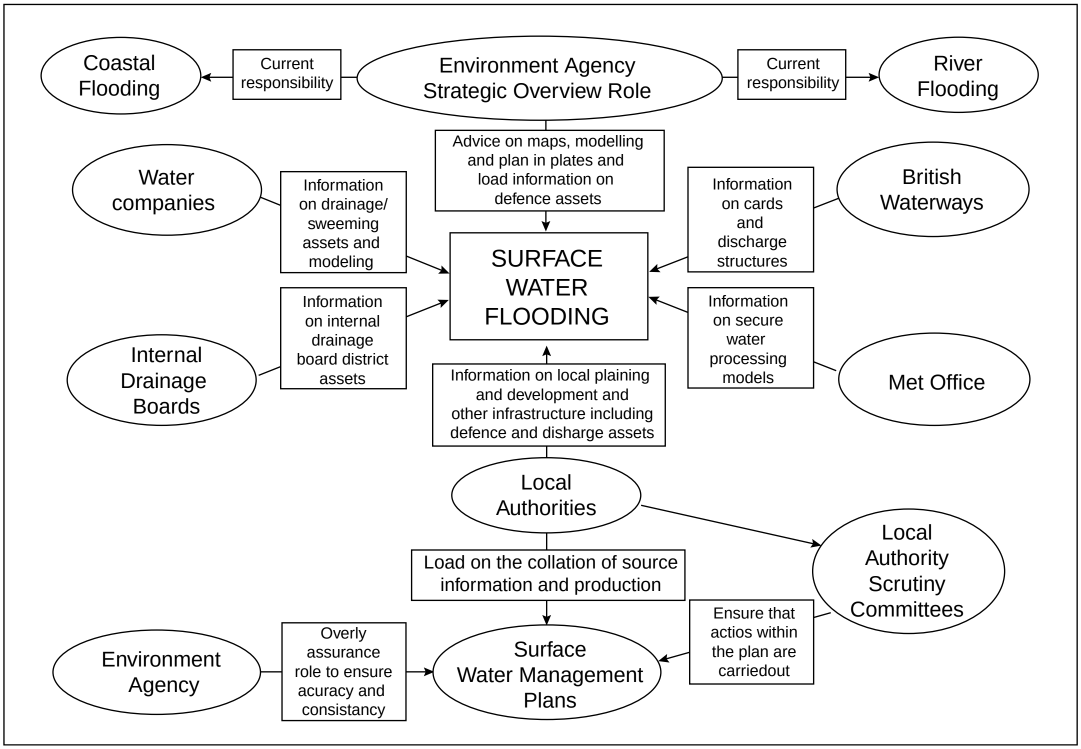

Fig. 13.1 Key stakeholders with responsibilities in flood risk management. From Pitt (2007).

of the performance of the integrated urban drainage system.

Planning is also a key regulatory driver, and new planning guidance, in the form of PPS25 (2006), sets out UK government policy on building and construction development and flood risk. CIRIA (Balmforth et al. 2006) outlined best practice advice for the design and management of urban sewerage and drainage systems to reduce the impacts that arise when the flows exceed the capacity of the system and of ways to adapt the existing system to accommodate the excess flows.

The EU Water Framework Directive has the specific aim to mitigate the effect of floods and droughts, and the EU Floods Directive (E 2007) required that preliminary flood risk assessments are completed to identify areas at risk, with the subsequent development of flood hazard and flood risk maps for areas at risk and the publication of flood risk management plans by December 2015. FRMRC has attempted to address many of the shortfalls in knowledge that impede the development of such risk maps, but first it is necessary to understand the mechanisms of the types and causes of urban flooding.

Urban drainage systems

In respect of urban flooding, reference is made to the major and minor components of the drainage system, where three components of a combined system are defined:

• **Minor system:** – the underground sewerage or drainage network that is designed for dry weather flow and storm flows of a given return period and duration. This includes gully inlets, local drainage, sewer pipes, culverted watercourses, CSO and storage structures, pumping stations and outfalls.

- **Overland major system:** – the urban surface that forms surface flowpaths at the time of rainfall events.
- **Major system:** – the above-ground network of preferential surface flowpaths through which storm water is conveyed during a flood event, including watercourses and rivers.

At the time of rainfall events these systems interact and, from an integrated urban flood management perspective, there is a need to understand this interaction and how the performance of the system responds to a range of urban flood drivers that cause different types of urban flooding. These are complex processes but a simplified line diagram (Fig. 13.2) highlights the way in which the systems are linked, whilst Figure 13.3, highlights the interaction of the above-ground surface flows (major system) and the below ground drainage system (minor system). The rainfall triggers surface runoff and, in the first instance, storm water enters the drainage network. When the drainage system becomes surcharged, water flows out of pipes back onto the surface, resulting in flooding. It is clear that, dependent on the flow conditions and time after the start of surcharge, the flow may be into the sewer from the catchment surface or out of the sewer onto the catchment surface. Here the role of gullies and manholes is particularly important. This leads to the consideration that there are many types of urban flooding.

Types of urban flooding

There are many types of urban flooding and each type results in a different type of surface water on the catchment surface. These include:

- **Pluvial flooding:** Rainfall in the urban area may cause flooding due to the fact that there are inadequate hydraulic access pathways to the underground sewer system or due to the fact that the pipes in the sewer system have a hydraulic capacity that is less than the flows that are generated by the rainfall runoff process. In the latter case the sewer pipes are hydraulically inadequate and this results in a back-up of flow and a 'surcharge' of the system. Such surcharge may result in the internal flooding of basements or external flooding of the catchment surface with the consequent flooding of properties.
- **Flooding of the urban area from surrounding catchments:** Such flooding occurs due to a rainfall event on the rural or peri-urban area that surrounds or is adjacent to the intra-urban area within the same catchment. If the flowpaths from these surrounding areas lead directly to the intra-urban area it is feasible for the surface runoff to flood the intra-urban area, in a similar way to floods caused by pluvial flooding. In the case where the slopes of the surrounding area are steep this type of flooding may be severe (flash floods).
- **Fluvial inundation from inland waterways:** The performance of the sewer system in the intra-urban area may be influenced by the performance of the fluvial drainage system in two ways:
  - Hindered performance due to a back-up of flow in the sewer system caused by enhanced fluvial flows that inundate the discharge outlets of the sewer system.
  - Inundation of the intra-urban catchment surface due to the failure, overtopping or by-passing of the flood defences of the fluvial system. This results in an inundation of the sewer system, which becomes full and subsequently inoperable due to extremely slack hydraulic gradients, often with ponding on the low-lying areas of the catchment surface.
- **Inundation in coastal regions:** Similar impacts to those observed in inland systems may be observed in the intra-urban areas adjacent to coastal and estuary environments where, similarly, the performance of the sewer system may be hindered by the height of surges or the overtopping or failure of coastal defences.
- **Asset performance, deterioration or failure:** Sewer flooding is also caused by the performance of assets and asset failure. In many cases such performance is governed by the condition and status of the assets and of the way in which they deteriorate. The primary processes that cause such flooding include:
  - blocked or restricted sewer outfalls and inlets;
  - sewer blockages and collapses;
  - presence of sewer sediments;
  - mechanical and electrical failure.

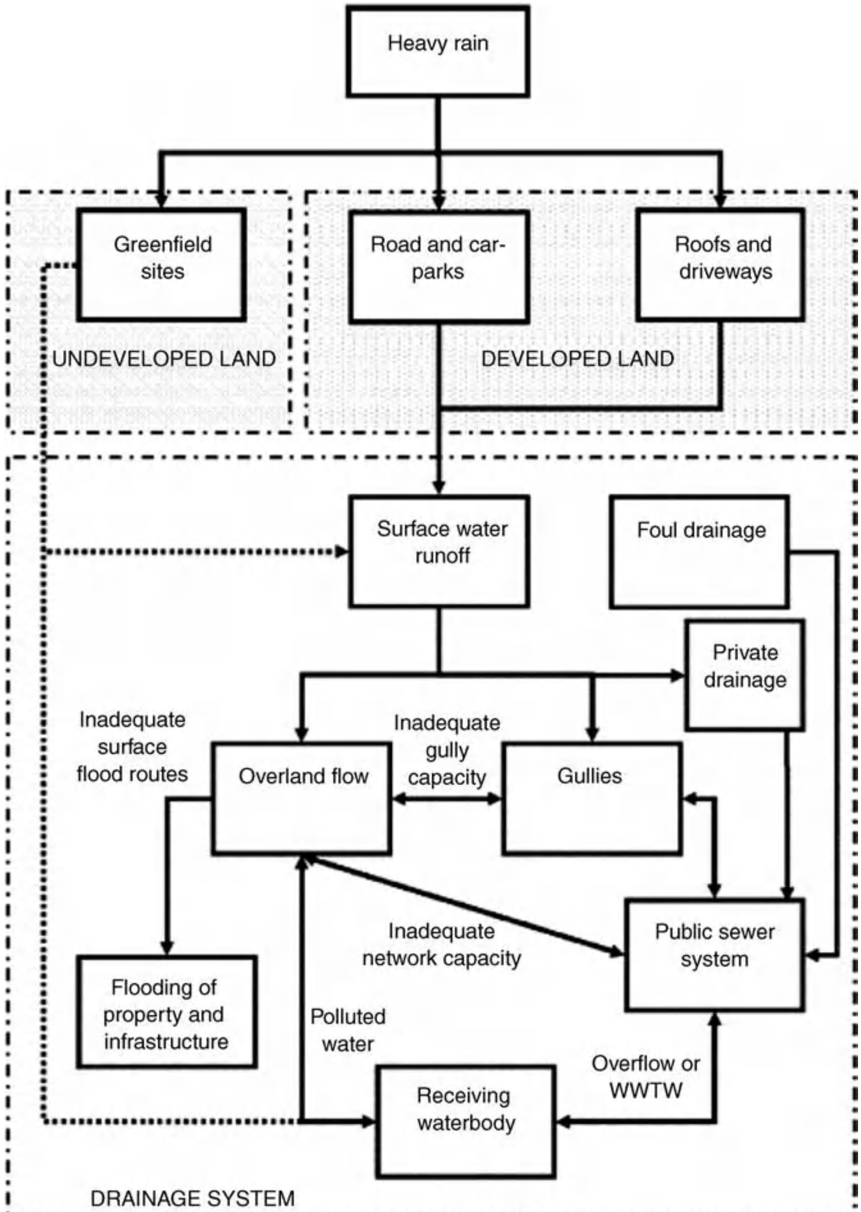

Fig. 13.2 Links between the major and minor urban drainage systems at the time of rainfall. After WaPUG (2009).

• **Groundwater flooding:** Flooding of the urban area from a high groundwater level.

• **Coastal flooding:** The flooding of urban areas in coastal towns and cities due to tidal surges and waves.

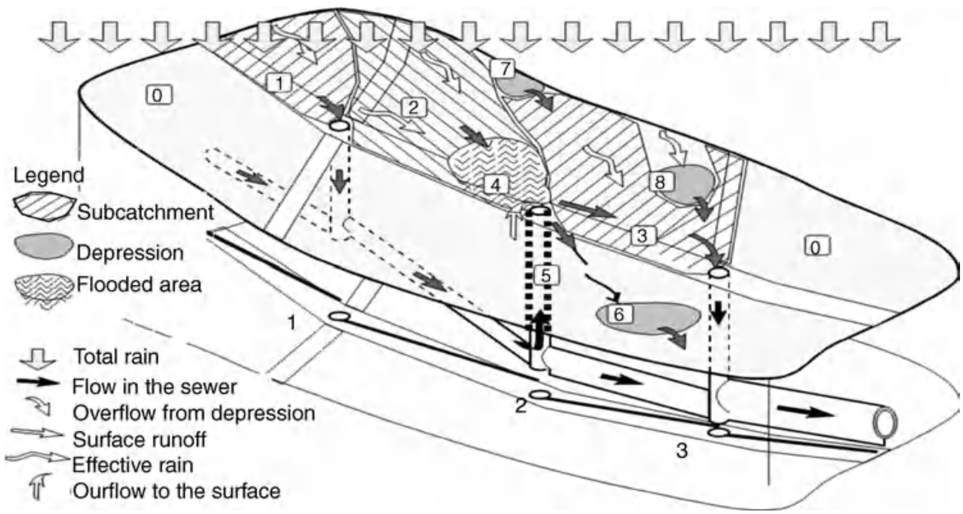

Fig. 13.3 The concept of interactions in the urban flooding process. From Djordjević *et al.* (1999).

• **Coincident flooding:** Coincident flooding occurs when two or more of the above flooding mechanisms occur simultaneously.

With reference to Figure 13.3, it is clear that there are a series of interactions between the different components and types of urban flooding. Subsequently, the latest research has focused on the development of integrated approaches to urban flood risk management.

FRMRC urban flood modelling aims

To meet the challenges of an improved understanding of urban flood risk, the Flood Risk Management Research Consortium (FRMRC) funded the development of an 'Integrated surface and subsurface interactive flooding model'. The specific aims of the research were as follows:

• To enhance existing overland surface flow and dynamic state hydraulic models for the underground drainage system that simulate flood events in the urban area (depth, velocity and volume).

• To develop a mathematical model to describe the hydraulic performance of the overland surface flow and the underground pipes in separate and combined sewer networks at the time of flood events. Specifically the model should take into account the key performance indicators for all types of urban flooding and how they change over the duration of a flood event.

• To present the integrated model in such a format that it may be used by practising engineers.

• To complete at least two case studies in collaboration with UK Water Industry Research (UKWIR).

FRMRC integrated model development

Historically, the design and performance of the below-ground drainage system and the above-ground major systems have been treated in isolation, with individual models applied to each component of the system. For underground drainage systems and in-bank river flows it is usual to apply a one-dimensional (1D) model as the flows in the system are laterally constrained within the pipe or river cross-section and the velocity of flow is in one direction. Established procedures for 1D modelling of sewer networks are outlined in the WaPUG Code of Practice for Hydraulic Modelling of Sewer Systems (WaPUG 2002), and details of appropriate models for rivers are outlined in Chapter 12. However, in practice, the direction of the overland flows within an urban area is not constrained and is very much influenced by the topography and the characteristics of the surface features and ground cover of the catchment surface. Hence, in a similar way to that in which river and floodplain systems are modelled (see Chapter 12), it is considered appropriate to utilize a two-dimensional (2D) model to describe the flow processes over the urban surface area. Both minor/major 1D/1D and 1D/2D models have been developed and tested as part of FRMRC with the aim to produce a geographical information system (GIS)-based tool for the analysis of surface runoff in urban areas during extreme wet weather.

For the 1D/1D case, use was made of the SIPSON model (Djordjević *et al.* 2005, 2007) supported by the overland flow module (Maksimović *et al.* 2009). The conceptual framework of the way in which the major and the minor systems are linked is shown in Figure 13.4. It is recognized that there are many water industry standard software models, for example Infoworks, MOUSE and SWMM, that may be used with confidence to predict the 1D flow in sewer systems. However, the SIPSON model was developed specifically to simulate the integrated performance of the above-ground surface flows and the flows in the below-ground drainage system. Similarly, the Urban Integrated Model (UIM) was developed as part of FRMRC to examine the interaction of 2D surface flows with the 1D below-ground sewer flow. The development of UIM (Chen *et al.* 2007) is described but it is also recognized that subsequent to the inception of the UIM model, other proprietary software houses have also developed models for such applications, for example Infoworks CS 2D. As a consequence, the final UKWIR case study has attempted to compare the flood prediction performance of the SIPSON/UIM model and the Infoworks software.

In summary, therefore, the FRMRC research considered here is divided into three sections. **Surface flow modelling**, which describes and illustrates the way in which the surface overland flow components of the model were developed, both in 1D and 2D, followed by details of the **Integrated surface/subsurface flow model** with a description of the numerical procedures and associated linkages between the surface/subsurface interactions. The final section of the chapter details examples of the application of the 1D/1D and 1D/2D models, where the application of the FRMRC outputs have been applied in **three UKWIR-funded case studies**.

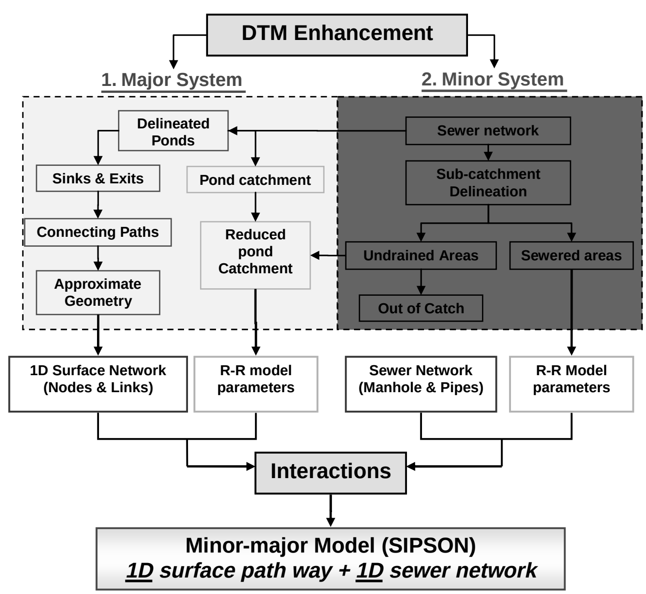

Fig. 13.4 Schematic of the model for the simulation of urban flood (1D/1D) interactions. DTM, digital terrain model, R-R Rainfall-Runoff.

Surface Flow Model

As a typical example, let us consider pluvial flooding of the urban surface. This is caused by the fact that the rainfall runoff on the catchment surface exceeds the capacity of the below-ground minor drainage system. This results in an overland flow that follows the urban surface flood pathways, which form part of the major system. Typically these pathways comprise roads and surface pathways that are linked to natural ground depressions and small watercourses. The approach to the modelling of overland flow in the urban environment caused by extreme rainfall was originally described by Prodanović (1998) and Djordjević (2001), but it is recognized that such pathways can transfer flow over significant distances with the consequence that flooding may occur at remote locations well away from the point at which the minor system capacity is exceeded. Similarly, surface runoff from adjacent areas that surround the urban area and that have no direct connection to the minor drainage system, may also result in urban flooding. The latest developments of this methodology are presented in Maksimović *et al*. (2009). Guidelines to manage such overland flows are outlined in CIRIA C635, Designing for Exceedance in Urban Drainage Systems (Balmforth *et al*. 2006), and it is clear that to accurately describe the path and the destination of the excess overland surface flows requires that there is a detailed representation of the nature and characteristics of the catchment surface such that the overland flood route and flood volume and velocity may be reliably represented. Within the context of urban flooding, this usually involves the identification of flood-vulnerable areas (mainly ponds), together with a definition of the preferential pathways linking the ponds. The overland flow may then be coupled with the below-ground drainage network, using a physically based model, to assess the interaction between the above- and below-ground flow components, as first suggested by Maksimović and Radojković (1986).

Today the usual approach is to make use of a GIS Digital Terrain Model (DTM) and/or a Digital Elevation Model (DEM). DTMs are frequently referred to as 'bare earth model', and these show the ground-level elevation without structures, buildings and vegetation, whereas the DEM includes these data (Leitão *et al*. 2006). Hence DTMs give a good representation of the contours and the natural flowpaths of the major system whereas the DEM may be used to highlight the surface features – buildings, walls, hedges, trees, etc. – that will interfere with the natural surface flowpaths. Hence DEMs have a better definition of any man-made flowpaths.

Preparation of DTM and DEM

An accurate description of urban flood risk requires accurate representation of the physical processes of overland surface flow, surface retention, and surface conveyance along preferential pathways. These require a high quality of terrain data, particularly in urban areas, where urban features such as buildings and streets need to be accurately described, both in plan view (x,y coordinates) and in the vertical presentation (z coordinate). Hence, the performance and reliability of the surface overland flow model are highly dependent on quality and resolution of the DTM/DEM. This is a function of the accuracy of the type of survey that is used to record the elevation data and the details of the catchment surface. Data may be obtained from many sources.

Maps

The UK's Ordnance Survey Mastermap data give a good representation of the surface features and provide the opportunity for visual inspection of potential flood flowpaths and ponds.

Light detection and ranging (LiDAR) data

Data are obtained by aerial survey and may be used to create a DTM or a DEM. There is a requirement to calibrate and validate the data, but for the purposes of urban flood modelling a vertical accuracy of ± 50 mm to ± 150 mm is desirable at a horizontal grid spacing of 0.5–1.0 m. Drive-by LiDAR, using instruments located in road vehicles, may also be used to improve the accuracy of local measurement where, for example, the elevation of kerb heights and local features are critical.

Conventional aerial LiDAR will fail to observe any local detail that cannot be seen from the air, for example the location of drop kerbs, the presence of covered passageways between buildings or of the potential flood paths under bridges. Some of these features may be identified using a fuzzy classification of the DEM (Evans 2008). It is considered essential that site visits are made to observe these features, particularly at locations where the DTM highlights surface depressions or major flood flowpaths.

Using fixed-winged aircraft it is currently feasible to record data at a resolution of 1 m horizontal and ± 150 mm vertical, compared with a resolution of 0.25 m horizontal and ± 30 mm vertical for rotary-winged aircraft.

Experience has shown that it can be useful to specify four layers of data to be delivered by the LiDAR survey contractors:

- the first pulse return DEM, which includes vegetation;
- the last pulse return DEM, which includes hard vegetation (large tree boles and dense hedge lines), buildings and solid artefacts;
- the DTM with all vegetation and artefacts and buildings removed;
- the DEM with buildings included.

The first three layers can be used to identify permeable and impermeable surfaces and the natural flood flowpaths, whereas the last three layers can be used to assess the impact of buildings, other artefacts and hard vegetation on flowpaths. However, it may be necessary at critical locations to improve the resolution of DEMs by means of drive-by LiDAR surveys, global positioning system (GPS) surveys or other methods.

Land and GPS surveys

These provide the most accurate method of obtaining both elevation and ground cover data, together with accurate measurements of any features that are likely to influence the surface flood flowpaths and flood-vulnerable areas. They also afford the opportunity to locate the gulley and manhole access points to the below-ground drainage system, and their condition, together with details of any walls, hedges and 'hidden flowpaths'.

It has to be recognized that small changes in urban surface topography can significantly change flowpaths, and it is recommended that site walkovers (Allitt *et al.* 2008), for example, are made such that further detailed information may be obtained and photographic records made. In addition to the road layout, gulley and manhole spacing, kerb heights and drop kerbs, buildings, etc., important surface features include both small and major embankments, bridges, retaining walls, culverts and open-access flowpaths through buildings.

Similarly, extra care should be taken where the catchments are flat as the plan area of flooded water and the flow depths may be large with low velocity. As a consequence, further increases in depth could result in a significant change in the flow route, for example when retaining walls are overtopped.

The FRMRC approach to develop the surface flow model

The approach was based on analysis using an accurate DTM/DEM and the creation of separate GIS layers to identify and define the flood-vulnerable areas (mainly surface ponds) and geometric characteristics of the preferential pathways through which the flood waves were routed over the catchment surface. To model urban flood flows requires that the water movement over the catchment surface is modelled by solving the appropriate mass and momentum (or energy) conservation equations. On the surface this includes the dynamics of the processes that occur in temporary surface retention (ponds) and of the flow across the urban catchment along preferential pathways. The ponds and pathways are mutually connected and multiple connections may exist with inlets to the underground sewer network. During a flood event these two networks interact, and when the sewer network is surcharged water can flow both into and out of the sewer network, as a function of the level of surcharge.

In FRMRC the surface runoff was routed by using the full dynamic St-Venant equations. This had the advantage that once the model had been calibrated, any changes in land use or physical characteristics could be represented by changing the relevant input files without compromising modelling integrity.

The following steps were undertaken to develop the surface flow components, after Boonya-aroonnet *et al.* (2007).

Identification of ponds and flood-vulnerable areas

In FRMRC the DEM raster image was used to analyse flood vulnerability and the entire dynamic processes. The enhanced DTM was firstly used to identify the location of the depressions or ponds. For each pond the stage, depth, volume and surface area relationships were established. Ponds may be isolated or mutually connected. When the ponds are full they overflow at a location termed the exit point (or points) and the spilled flow enters a preferential flow path on the catchment surface. Frequently ponds are nested within larger ponds; for example, at shallow depths there may be two isolated ponds but at larger depths these are mutually connected to form one large pond. Flood-vulnerable areas are usually located in local depressions (ponds) but they may also occur within pathways. The search algorithm examines the elevation data to extract the lowest points in the DEM, and these are flagged as potential ponds. Based on the DEM, the pond boundary for each low point is delineated and the natural exit point is identified by a review of cell elevation, as detailed in Figure 13.5. The exit point represents the upstream end of the overland flow pathway and the hydraulic characteristics of the exit point are used to determine the discharge capacity of the outlet from the pond.

Connectivity analysis

The DEM, which also includes urban (man-made) features such as streets and buildings, was then used to identify the surface pathways for overland flow. These pathways connect the previously identified ponds in order to form a 'surface flow network'. The upstream flowpaths start at the exit point(s) of the ponds or issue onto the catchment surface from the below-ground system at manholes, whilst the downstream end of the pathway was defined as the entry to another pond, the flow entry into the sewer via the manhole or as an outflow from the catchment. As detailed in Figure 13.6, overland flow may be considered to accumulate in depressions, and once the top level of depression is reached, it will either overflow

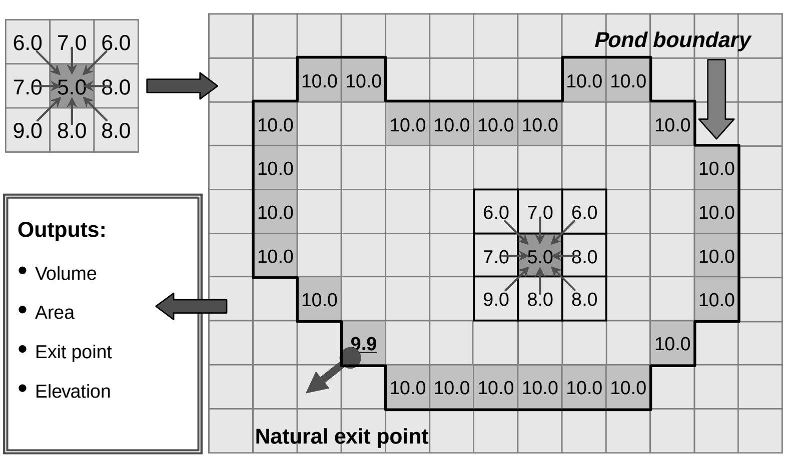

Fig. 13.5 The pond delineation (numbers in cells are elevations). After Boonya-aroonnet *et al.* (2007).

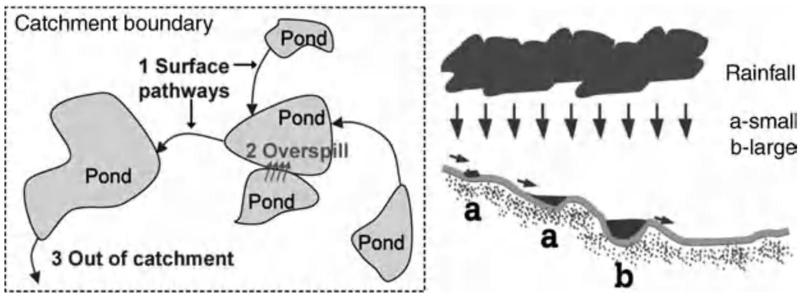

Fig. 13.6 Pond connectivity calculated from a digital elevation model (DEM) containing small (a) and large (b) depressions.

directly to an adjacent pond or will flow along a preferential pathway until it reaches another depression or an inlet to the sewer network. The connectivity algorithm used in FRMRC was developed by AUDACIOUS (2005), and utilized the 'rolling ball' technique to identify surface pathways and the flow between adjacent ponds. This technique was subsequently enhanced using a 'sliding ball' technique to cope with the areas in which 'rolling ball' comes to rest ('gets stuck'). The analysis commences at the exit point of the upstream pond and identifies the preferential flood pathway, based on terrain slope and the presence of buildings and other features of the urban catchment surface.

Automatic subcatchment delineation

Sub-areas that contributed flow to individual drainage elements are established from the DTM and are introduced to the model via the nodes and/or links as appropriate. The sub-areas may change according to the magnitude of the event; for example, there may be additional runoff from permeable areas, or an increased depth of flow in the pathway may result in a new route for the surface flow, for example the formation of a bifurcation.

The catchment surface is partitioned into smaller areas, using an automatic subcatchment delineation routine. Each subcatchment drains to a single network node. The procedure is based on the elevation of the DEM and the land cover and use, defined by the coordinates of the nodes (manholes and/or surface ponds) or links (pipes and/or surface pathways). The process is illustrated in Figure 13.7.

The process was first developed by Prodanović (1998) and the procedure takes into account the variability of flow angles over different types of cover (fabric or canopy), and the presence of artificial or man-made objects (streets, buildings). Below a selected threshold the slope of the terrain was considered horizontal.

Subcatchment delineation for sewered areas and definition of undrained areas

Subcatchment delineation is also used to identify the contributing area for each pipe in the sewer network. Figure 13.8 illustrates the methodology based on the use of the links (and not the nodes) to delineate the surface. Figure 13.8 also highlights that some areas, termed 'undrained areas', are not directly linked to any pipe's inlet. For small storms these areas may be considered to be unimportant as they contribute little runoff, but in extreme events the runoff from such areas can be significant and hence these areas should be designated 'undrained ponds'. An algorithm was developed in FRMRC to do this such that it was not possible to double count any areas common to both a pond and a subcatchment, as detailed in Figure 13.9.

Estimation of pathways and pathway geometry

The DTM is then used to establish suitable shapes representative of the channel cross-sections that

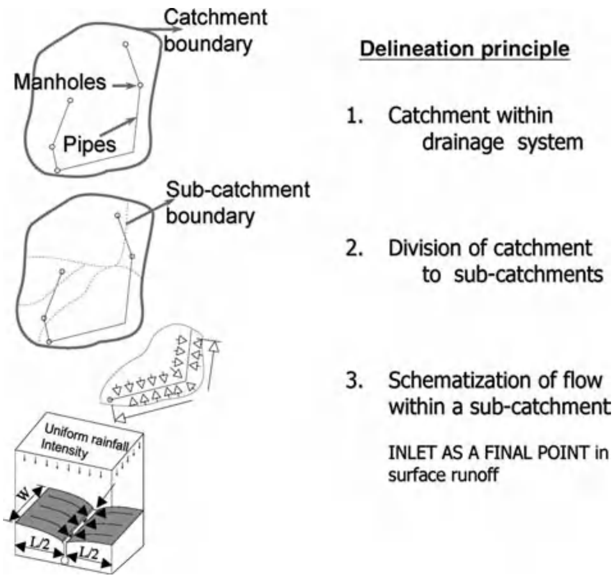

Fig. 13.7 Subcatchment delineation. L = catchment length, W = catchment width. (See colour version of this figure in Colour Plate section)

are used to define the drainage capacity of the surface flowpaths. Modelling of flow in surface pathways requires the following information:
• the geometry of the open-channel drainage on the surface;
• upstream/downstream elevations;
• roughness;
• the actual length along the pathway between two ponds or surface nodes.

The overall process is presented in Figure 13.10. The algorithm uses the preferential flowpaths derived from the DEM and constructs channel cross-sections equidistant along the length of each pathway (Fig. 13.10b). The DEM is also used to estimate the surface area of each cross-section, which is then transformed into (represented by) a trapezoidal or natural section. This channel is then used to route the flood flow over the catchment surface by taking into account the average flow areas at different depths along the length of each pathway (so-called 'stage-flow area' curve). These, together with the values of the average channel slope (based on the upstream and downstream channel elevation and the length of

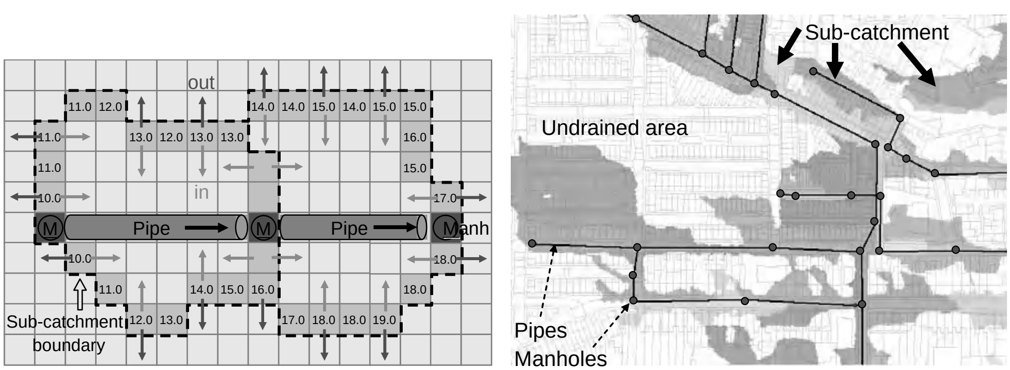

Fig. 13.8 Link based approach to determine sub-catchment delineation. (See colour version of this figure in Colour Plate section)

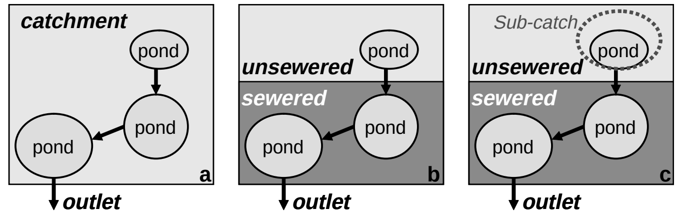

Fig. 13.9 Overlaying ponds with 'sewered' areas to find 'unsewered' or 'undrained' ponds and their contributing areas.

channel section) and bed roughness, extracted from the DTM, are exported to the model as inputs for simulation.

Model output

A typical output from the model that was developed is shown in Figure 13.11. This identifies the flow paths (blue) and the location of the ponds (yellow). The surface water system is shown in green, the combined sewer system is in red with the node manholes as small black circles, and the gullies are identified as dots by the kerb lines. Once the surface flow model is created it can be combined with the model of the subsurface system and rainfall-runoff (flooding) simulations may then be carried out. The physical processes of how these system components interact are described in the next section.

Integrated Surface/Subsurface Flow Model

As shown in Figure 13.4 the SIPSON model (Djordjević *et al.* 2005) was used to integrate the surface overland flow model with the below-ground drainage model. The surface and subsurface networks were physically linked at the manholes and road gullies, as shown in Figure 13.12. Here there are two possible modes of operation.

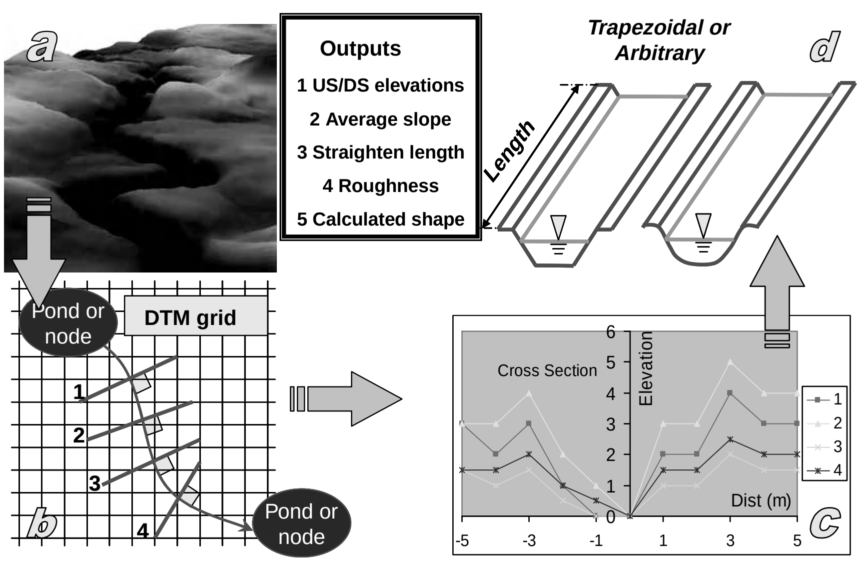

Fig. 13.10 Estimation of pathways geometry. (a) A 3D digital terrain model (DTM) showing identified flow path (U/S upstream, D/S downstream). (b) A number of cross-sectional lines drawn perpendicularly to the path. (c) The arbitrary shapes of cross-sections plotted as found from the DTM. (d) Averaged output with two choices, trapezoidal or actual shapes. (See colour version of this figure in Colour Plate section)

Fig. 13.11 Ponds and flowpaths derived from the digital elevation model (DEM). Flowpaths in blue; location of ponds in yellow; surface water system in green; combined sewer system in red; node manholes as small black circles; gullies identified as dots by the kerb lines. (See colour version of this figure in Colour Plate section)

Firstly, water from the pond or the preferential flowpath may enter the sewer via the manhole, or conversely, the flow in the sewer system may become surcharged and flows may exit from the manhole onto the catchment surface or into a pond. After the storm the flood waters re-enter the drainage system through manholes. The interaction is dynamic and occurs over the complete duration of an event. The interactions between surface and subsurface networks are enabled by virtual weirs or orifices or combinations (termed 'equivalent inlets'), which connect the surface network nodes (ponds, street junctions, point-type junctions) with subsurface network nodes (manholes). Fuller details may be found in Leandro *et al.* (2007).

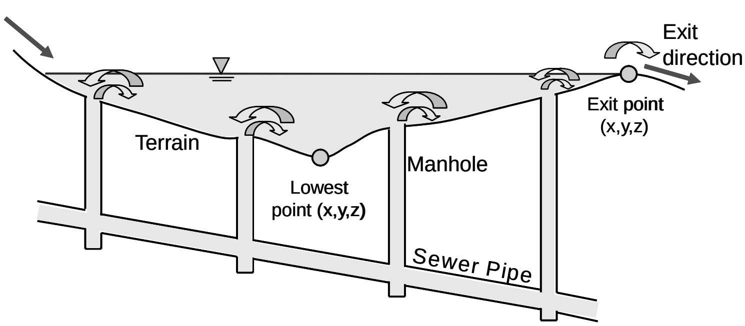

Fig. 13.12 The interaction between manholes and a surface pond.

The model solves all flow equations simultaneously such that the solution procedure does not distinguish between the flows on the catchment surface and the flows in the below-ground drainage system. The only distinction between the surface flow paths and the underground pipes relates to differences in the characteristics of the individual elements; for example, sewer pipes have closed cross-sections (circular, egg-shaped, etc.) whereas surface links are open channels, typically with irregular geometry, which can be approximated by a regular cross-section (e.g. trapezoidal). Hence, any pipe, channel, inlet, weir or pump is seen by the model as a link within one integrated network.

Governing equations and notes on specific parameters

A mathematical model of flow in a network consists of a system of equations that describe all forms of free-surface and surcharged flows.

Continuity at nodes

At nodes, the continuity equation can be written as:

$$F\frac{dZ}{dt} = q + \sum_{m=1}^{M} \pm Q_m \qquad (13.1)$$

where F = node horizontal area, Z = water level at the node, t = time, q = external inflow to the node (surface runoff, waste water, etc.), M = number of links joining the node and Q_m = discharges flowing from the link to the node or vice versa.

Free surface flow in links

One-dimensional free-surface flow in a pipe or a channel may be described by the complete St-Venant equations, which can be written in the form:

$$\frac{\partial z}{\partial t} + \frac{1}{B}\frac{\partial Q}{\partial x} = 0 \qquad (13.2)$$

$$\frac{\partial Q}{\partial t} + \frac{\partial}{\partial x}\left(\frac{Q^2}{A}\right) + gA\left(\frac{\partial z}{\partial x} + S_f\right) = 0 \qquad (13.3)$$

where z = cross-sectional water level, B = water surface width, Q = discharge, x = space coordinate, A = cross-sectional area, g = gravitational constant and S_f = friction slope.

Surface/subsurface links

There are three basic cases of flow through surface/subsurface links, as shown in Figure 13.13. When a hydraulic head in the manhole is below the ground level (Fig. 13.13a), it does not influence the flow through the inlet. Thus the inflow can be described by either a weir equation for shallow depths or an orifice equation when the area of opening of the inlet is submerged.

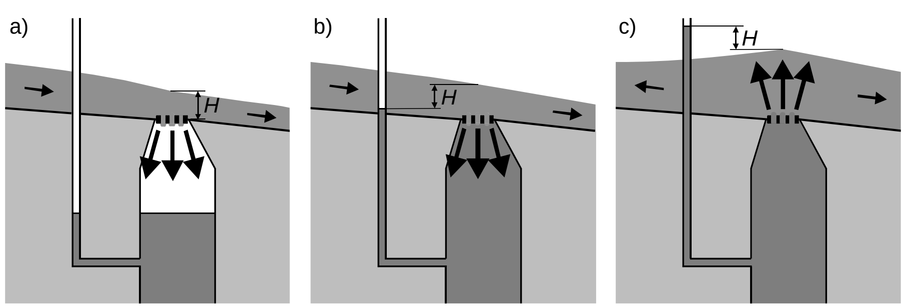

Fig. 13.13 Basic cases of flow through equivalent inlet: (a) free inflow, inlet as a weir (H, water depth on the surface); (b) submerged inflow, inlet as an orifice (H, difference between water level on the surface and hydraulic head in the manhole); (c) outflow (H, difference between hydraulic head in the manhole and water level on the surface).

When a head in the manhole is between the ground level and the water level on the surface (Fig. 13.13b) the water still flows from the surface to the pipes, and may best be represented using an orifice equation. However, it is stressed that the selection of the appropriate discharge coefficient is uncertain.

Finally, if the below-ground system becomes surcharged and the head in the manhole is greater than that created by the water level on the street (Fig. 13.13c), water from the below-ground system will issue from the manhole onto the street surface. Here, the most appropriate formula to describe the flow out of the inlet is that of an orifice equation, but again it has to be recognized that the selection of the appropriate discharge coefficient is uncertain.

What is clear is that the mathematical equations used to describe the change in flow regime, from (a) to (b) to (c) and vice versa, and their discharge coefficients, have to form a continuum with a smooth transition in the head discharge relationships between inflow and outflow values and vice versa. Research is ongoing to address this issue.

Numerical solution procedure for the below-ground drainage system

The numerical procedure described here utilizes the general algorithm for solving finite difference problems originally introduced by Friazinov (1970). It is based on the idea of temporary elimination of variables at internal cross-sections and thus reducing all equations to a system of unknown water levels at network nodes. This is an elegant algorithm, and, combined with different numerical methods and various matrix solvers, has been used in several commercial models, for example, InfoWorks CS and MOUSE.

The St-Venant equations are solved by a variant of the Preissmann implicit finite-difference method where:

$$\frac{\partial f}{\partial t} \approx \psi \frac{(f_{i+1}^{j+1} - f_{i+1}^{j})}{\Delta t} + (1-\psi)\frac{(f_i^{j+1} - f_i^{j})}{\Delta t} \qquad (13.4)$$

$$\frac{\partial f}{\partial x} \approx \theta \frac{(f_{i+1}^{j+1} - f_i^{j+1})}{\Delta x} + (1-\theta)\frac{(f_{i+1}^{j} - f_i^{j})}{\Delta x} \qquad (13.5)$$

$$f \approx \theta[\psi f_{i+1}^{j+1} + (1-\psi) f_i^{j+1}] + (1-\theta)[\psi f_{i+1}^{j} + (1-\psi) f_i^{j}] \qquad (13.6)$$

where f = any function, ψ, θ = spatial and temporal weighting coefficients, respectively, and i, j = space and time indices, respectively. Commonly, $\psi = 0.5$ and $\theta = 0.67$. Where f is a product or a ratio of variables there is no separation and the products/ratios are discretized as such. Where any variable φ is in front of the differentiation operator, the discretization is as follows:

$$\phi \frac{\partial f}{\partial x} \approx \theta \frac{(\phi_i^{j+1} + \phi_{i+1}^{j+1})}{2} \frac{(f_{i+1}^{j+1} - f_i^{j+1})}{\Delta x} + (1-\theta)\frac{(\phi_i^{j} + \phi_{i+1}^{j})}{2}\frac{(f_{i+1}^{j} - f_i^{j})}{\Delta x} \qquad (13.7)$$

Substitution of Equations 13.4 to 13.7 into Equations 13.2 and 13.3 and linearization leads to:

$$\mathrm{a}_i Q_i^{j+1} + \mathrm{b}_i z_i^{j+1} + \mathrm{c}_i Q_{i+1}^{j+1} + \mathrm{d}_i z_{i+1}^{j+1} = \mathrm{e}_i \qquad (13.8)$$

$$\mathrm{a}' Q_i^{j+1} + b'_i z^{j+1} + c'_i Q_{i+1}^{j+1} + d'_i z_{i+1}^{j+1} = \mathrm{e}'_i \qquad (13.9)$$

where $\mathrm{a}_i, \mathrm{b}_i, \ldots, \mathrm{e}_i$ = abbreviations, most of which include variables Q^{j+1} or z^{j+1}. Equations 13.8 and 13.9 for all subreaches of a single pipe/channel form the system of algebraic equations (for $i = 1, 2, \ldots, N-1$, where N = number of cross-sections; see Fig. 13.14).

By eliminating the unknowns at internal cross-sections (from $i = 2$ to $i = N - 1$), this system of $2N - 2$ equations is reduced to the equivalent system of two equations:

$$Q_1^{j+1} = \mathrm{f}_1 z_1^{j+1} + \mathrm{g}_1 z_N^{j+1} + \mathrm{h}_1 \qquad (13.10)$$

$$Q_N^{j+1} = \mathrm{f}_N z_1^{j+1} + \mathrm{g}_N z_N^{j+1} + \mathrm{h}_N \qquad (13.11)$$

where $\mathrm{f}_1, \mathrm{g}_1 \ldots, \mathrm{h}_N$ = abbreviations. Substitution of free-outflow conditions and/or the energy

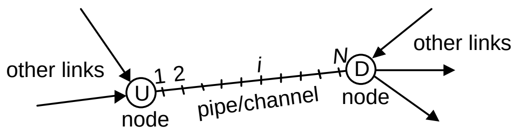

Fig. 13.14 Network elements. U, upstream network node; D, downstream network node; *i*, *N*, index and number of computational cross-section, respectively.

conservation equation into Equations 13.10 and 13.11 transforms those into:

$$Q_1^{j+1} = f'_1 Z_v^{j+1} + g'_1 Z_D^{j+1} + \mathrm{h}_1 \qquad (13.12)$$

$$Q_N^{j+1} = f'_N Z_v^{j+1} + g'_N Z_D^{j+1} + \mathrm{h}'_N \qquad (13.13)$$

where indices U and D denote network nodes at upstream and downstream channel ends, respectively (Fig. 13.14). After possibly some linearization, relationships for other link types (weirs, pumps) can be expressed in the same form as well.

For each node, if $F(Z) > 0$, Equation 13.1 is solved by the Euler modified method:

$$Z^{j+1} = Z^j + \frac{\Delta t}{2}\left(\frac{\varphi^j}{F^j} + \frac{\varphi^{j+1}}{F^{j+1}}\right) \qquad (13.14)$$

where φ = righthand side of Equation 13.1. For point-type junctions (where $F = 0$), Equation 13.1 reduces to $\varphi^{j+1} = 0$. Substitution of Equations 13.12 and 13.13 for all joining links into Equation 13.14, and doing so for all the nodes, leads to a system that can be written in matrix form:

$$\mathbf{p} \cdot \mathbf{z} = \mathbf{q} \qquad (13.15)$$

where $\mathbf{Z}$ = a vector containing unknown water levels at network nodes and $\mathbf{p},\mathbf{q}$ = coefficient matrices. As node matrix $\mathbf{p}$ can be rather large but with a large number of zero terms, it may be banded into a row-indexed sparse storage form. Then the system (Equation 13.15) is solved by the conjugate gradient method considering the function:

$$\Phi(\mathrm{Z}) \equiv \frac{1}{2}|\mathbf{p} \cdot \mathbf{Z} - \mathbf{q}|^2 \qquad (13.16)$$

and minimizing the expression $\Phi(\mathbf{Z} + \lambda\mathbf{u})$, where vector $\mathbf{u}$ is the gradient of function Φ from the previous iteration, $\mathbf{u} = \nabla\Phi(\mathbf{Z}^*)$, and scalar $\lambda = -\mathbf{u} \cdot \nabla\Phi / |\mathbf{p} \cdot \mathbf{u}|^2$.

Once the system (Equation 13.15) is solved, the discharge at any link end can be determined from Equations 13.12 or 13.13. For pipe/channel links, where flow direction is from the node to a link, the discharge thus obtained becomes the boundary condition at that channel end. Otherwise, the water level calculated from the energy conservation equation or from the free-outflow condition becomes the boundary condition.

Finally, these boundary conditions are added to the system of Equations 13.8 and 13.9, which is rearranged to a tridiagonal form and solved using a double-sweep technique and the Newton–Raphson method. This system of equations forms one global iteration within one time step, and since linearization is applied at several stages within the process, typically two to three global iterations are sufficient to meet the convergence criteria.

Pressurized flow is simulated by the open-slot method and supercritical flow is treated using Havno's approximations. These techniques are also used in standard commercial packages such as InfoWorks CS, MOUSE and others.

The urban inundation model (UIM)

The UIM has been developed as a 2D non-inertia model derived from the St-Venant equations, with the inertial terms neglected by assuming the acceleration terms of the water flow over the land surface are relatively small compared to gravitation and friction terms. The continuity and momentum equations are written as:

$$\frac{\partial d}{\partial t} + \frac{\partial ud}{\partial x} + \frac{\partial vd}{\partial y} = q \qquad (13.17)$$

$$\frac{\partial(d+z)}{\partial x} + \frac{n^2 u\sqrt{u^2+v^2}}{d} = 0 \qquad (13.18)$$

$$\frac{\partial(d+z)}{\partial y} + \frac{n^2 v\sqrt{u^2+v^2}}{d^{4/3}} = 0 \qquad (13.19)$$

where d is the water depth (in metres); u and v are the velocity components in the x and y directions, respectively (m/s); z is the surface elevation (m); and q is the rate of water entering or leaving ground surface per unit area, including the excess rainfall, the upstream catchments inflows, the influent and effluent of sewer networks, and the overland flow.

The computation of 2D overland flow is time-consuming. To speed up the simulations an adaptive time step has been used, whereby the time step is adjusted automatically based on the Courant stability criterion (Yu and Lane 2006) such that the largest time step that ensures numerical stability is selected.

The time step used in the SIPSON 1D below-ground model was linked to UIM and hence the default and upper bound of time steps in the UIM, Δt_0, were made the same as in the SIPSON model, ΔT, i.e., $\Delta t_0 = \Delta T$. At the end of each computational time step, the Courant condition was checked based on the latest calculated water depth and velocities, where:

$$\Delta t'_{m+1} \leq \frac{\Delta x}{\sqrt{gd_m + u_m}} \tag{13.20}$$

$$\Delta t'_{m+1} \leq \frac{\Delta y}{\sqrt{gd_m + v_m}} \tag{13.21}$$

where m is the index of the time step; $\Delta t'_{m+1}$ is the estimated time step length (in seconds) used in UIM for the $m+1^{\text{th}}$ step; $d_m = h_m - z$ is the water depth (m) of the computing grid at the m^{th} step; and u_m and v_m are velocity components (m/s) along x and y directions, respectively. If conditions are satisfied the value of Δt_0 was selected for the next time step.

Model linkage

The flow dynamics are simulated using distinct below- and above-ground models, executed individually, that are coupled and linked by exchanging information obtained at manholes. The inlets are often covered by grates, and this adds further complexity in respect of the selection of the appropriate discharge coefficients.

Interacting discharge The interacting discharge into and out of the manholes was calculated using a series of head discharge relationships based on the water level difference between that in the sewer network and the depth of the overland flow on the surface. The upstream and downstream levels for determining discharge were defined as $h_U = \max\{H, h\}$ and $h_D = \min\{H, h\}$, respectively, where H is the hydraulic head (m) at node and h is the water surface elevation (m) on the overland grid. The hydraulic performance of the system inlet was defined by one of three equations as a function of the relative magnitude of the flow depth on the surface and the water level in the below-ground drainage system: a free weir, a submerged weir and an orifice (Leandro *et al.* 2007).

Free weir linkage The free weir equation is adopted when the crest elevation z_{cyest} is between the values of the upstream water level h_U and the downstream water level h_D, as shown in Figure 13.15. The discharge is calculated by using Equation 13.22:

$$Q = sign[H-h]c_w w\sqrt{2g}(h_U - z_{cyest})^{\frac{3}{2}} \tag{13.22}$$

where Q is the interacting discharge (m^3/s), whose positive value means surcharge flow from sewer toward overland and negative value means drainage flow from surface into sewer; c_w is the weir discharge coefficient; w is the assumed weir crest width (m); and g is the gravitational acceleration (m/s^2).

Submerged weir linkage The submerged weir equation is used (Fig. 13.16) when both water levels at node and overland grid are greater than the crest elevation, and the upstream water depth above the crest, $(h_U - z_{cyest})$, is less than $\frac{c_o A}{c_w w}$, where A is the node area (m^2). Equation 13.23 is employed for determining the interacting discharge:

$$Q = sign[H-h]c_w w\sqrt{2g}(h_U - z_{cyest})(h_U - h_D)^{\frac{1}{2}} \tag{13.23}$$

Orifice linkage The node is considered fully submerged (Fig. 13.16) when the upstream water

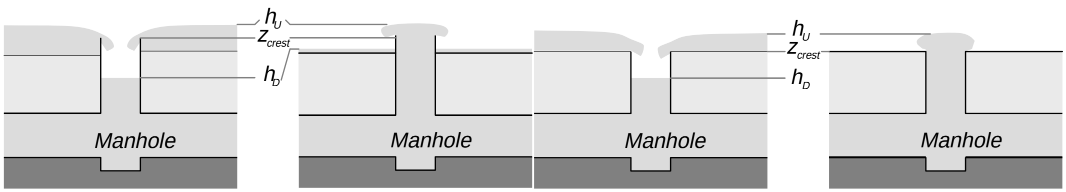

Fig. 13.15 Free weir linkages. h_U, upstream water level; h_D, downstream water level; z_{cyest}, crest elevation.

depth above the crest $(h_U - z_{cyest})$ is greater than $\frac{c_o A}{c_w W}$ for the submerged weir linkages. The orifice equation is used for calculating the interacting discharge:

$$Q = sign[H-h]c_o A\sqrt{2g}(h_U - h_D)^{\frac{1}{2}} \qquad (13.24)$$

where c_o is the orifice discharge coefficient.

Timing synchronization

The flow dynamics in drainage networks and overland surface are solved by two separate models using different time steps. Hence, to correctly link the models the synchronization of time is extremely important. This is particularly so as the 2D simulation uses a variable time step. The process of synchronization is shown in Figure 13.17.

Equations 4.20 and 4.21 provide an estimate of time step $\Delta t'_{m+1}$, and the next time at which the two models are synchronized is given by Equation 13.25:

$$\Delta t_{m+1} = \max\left\{(T_{syn_l} + \Delta T - \sum_{i=1}^{m} \Delta t_i), \Delta t'_{m+1}\right\} \qquad (13.25)$$

where Δt_{m+1} is the time step size (s) used for the $m+1^{th}$ step; T_{syn_l} is the time of the previous (l^{th}) synchronization (s); $\sum_{i=1}^{m} \Delta t_i$ is the total length of time steps (s) after m steps of calculations in UIM; $T_{syn_l} + \Delta T - \sum_{i=1}^{m} \Delta t_i$ is the time left (s) before the next (l + 1^{th}) synchronization.

The above equations were used to successfully link the 1D SIPSON and the 2D UIM modelling approaches to create a fully integrated above-ground surface flows and below-ground drainage system. Fuller details of the UIM model were reported by Chen *et al.* (2007).

Case Studies

Comparison of 1D/1D and 1D/2D Models at Keighley

The 1D SIPSON model and the 2D Urban Inundation Model (UIM) were coupled in a study to compare the performance of the 1D/1D and 2D/1D models developed as part of FRMRC. The case study area shown in Figure 13.18 is located near Stockbridge in Keighley, West Yorkshire, and was used to test the applicability of the models and the modelling process.

The study area had an overall catchment area of approximately 3.5 km^2, with the below-ground drainage system comprising 45 main sewer pipes

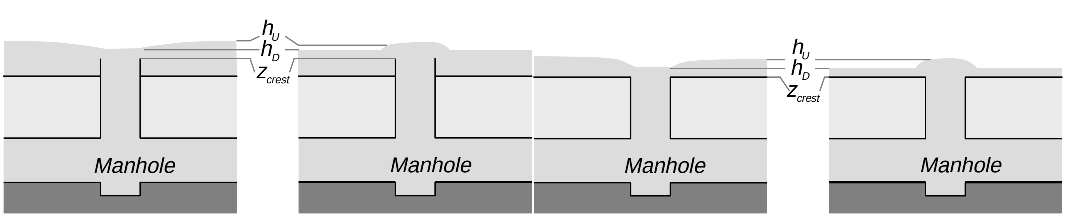

Fig. 13.16 Submerged weir $h_U \leq A/w$ or orifice $h_U > A/w$ linkages. A, node area; w, assumed weir width; h_U, upstream water level; h_D, downstream water level; z_{cyest}, crest elevation.

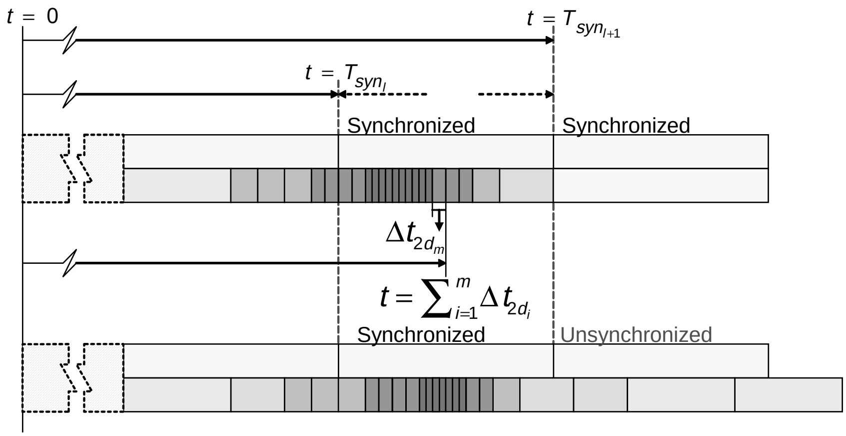

Fig. 13.17 Synchronized and unsynchronized timing between 1D and 2D models. See text for explanation of terms.

of total length of 6.0 km. The boundaries to the north and east of the catchment were set by the River Aire and one of its tributaries, and by a railway to the south and a major road to the west.

The DEM with buildings for the study area was constructed using LiDAR data at a 2×2 m grid spacing, as shown in Figure 13.19. The overall sewer network is also shown in Figure 13.19 but for the purposes of the comparative study a section of the system, with catchment area 0.2 km^2 containing 74 pipes of total length 3.0 km, also shown on Figure 13.19, was the subject of the case study.

The 2D surface terrain was schematized as a series of ponds connected by pathways in the 1D/

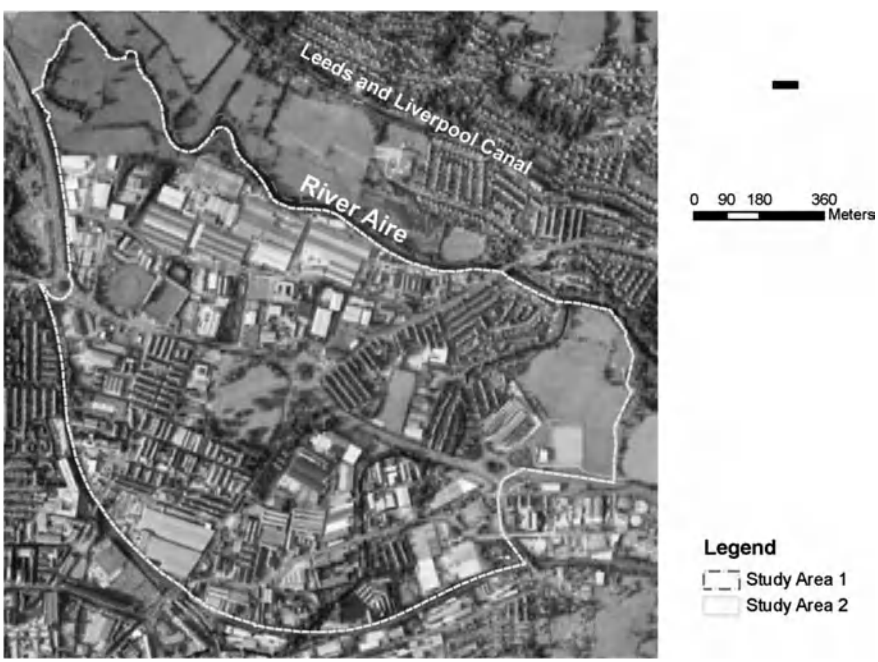

Fig. 13.18 Aerial photo of the study areas: Stockbridge in Keighley, West Yorkshire. (See colour version of this figure in Colour Plate section)

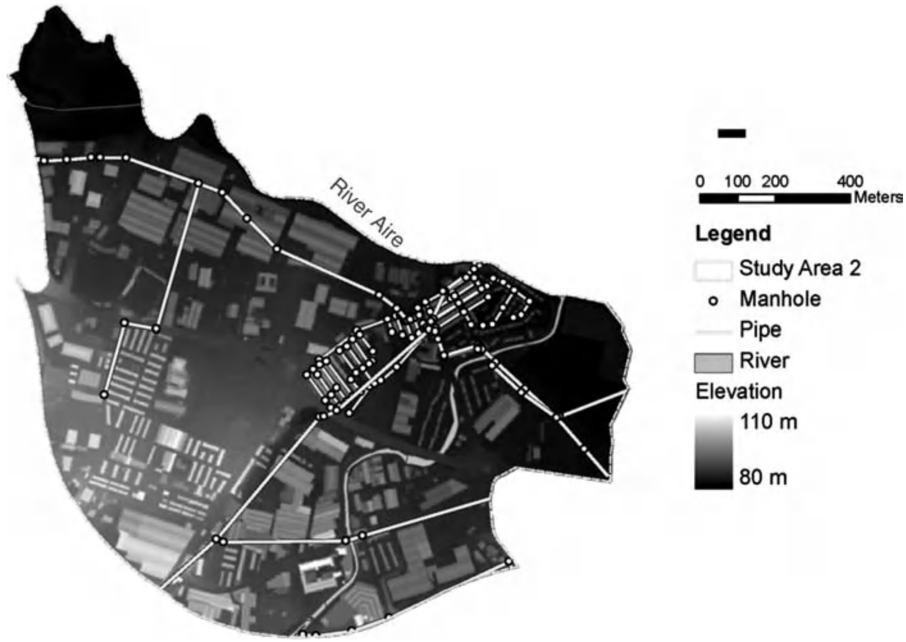

Fig. 13.19 The drainage system and the digital elevation model (DEM) of the study area. (See colour version of this figure in Colour Plate section)

1D model, and the same DEM was used in the 1D/2D modelling. There were 34 pathways and 22 ponds generated as the surface network, and 10 Multiple-Linking-Elements (MLEs) were used to describe the discharge coefficient(s) at manholes (Leandro *et al*. 2007). Seven weir equations were used as linkages between the ponds and manholes, the ponds and ponds, or the ponds and pathways, respectively.

An arbitrarily selected one-hour rainfall event of 200-year return period with 52 mm/h constant intensity was applied to the study area, and the outfall to the River Aire was represented as a free outlet. The results simulated by the 1D/1D and 1D/2D models for flood extent are shown in Figure 13.20. These figures highlight that the major flood extents were similar for both model outputs. The 1D/1D model produced a smaller flood area but with a greater flood depth due to the inundation being confined by the ponds. In the case of the 2D representation more opportunity is afforded for the overland flow to move over the terrain and this leads to wider flood extents and smaller average depths. However, a close examination of the results highlighted that, for the 2D model, at some local points the flow flood depth was greater than that predicted by the 1D model. This again emphasizes the value of the 2D terrain model in that within the 1D framework only the average ground elevation is considered for each grid cell.

This case study has highlighted that it is feasible to utilize the 1D/1D and 1D/2D modelling approaches to predict flood extent; a much fuller description of the application and calibration of the models is given by Leandro *et al*. (2009).

UKWIR demonstration projects

The next phase of the FRMRC urban flood research was to complete three UKWIR funded case studies to trial the research software in an industry environment. The intention of the studies was to enhance and improve understanding of the way in

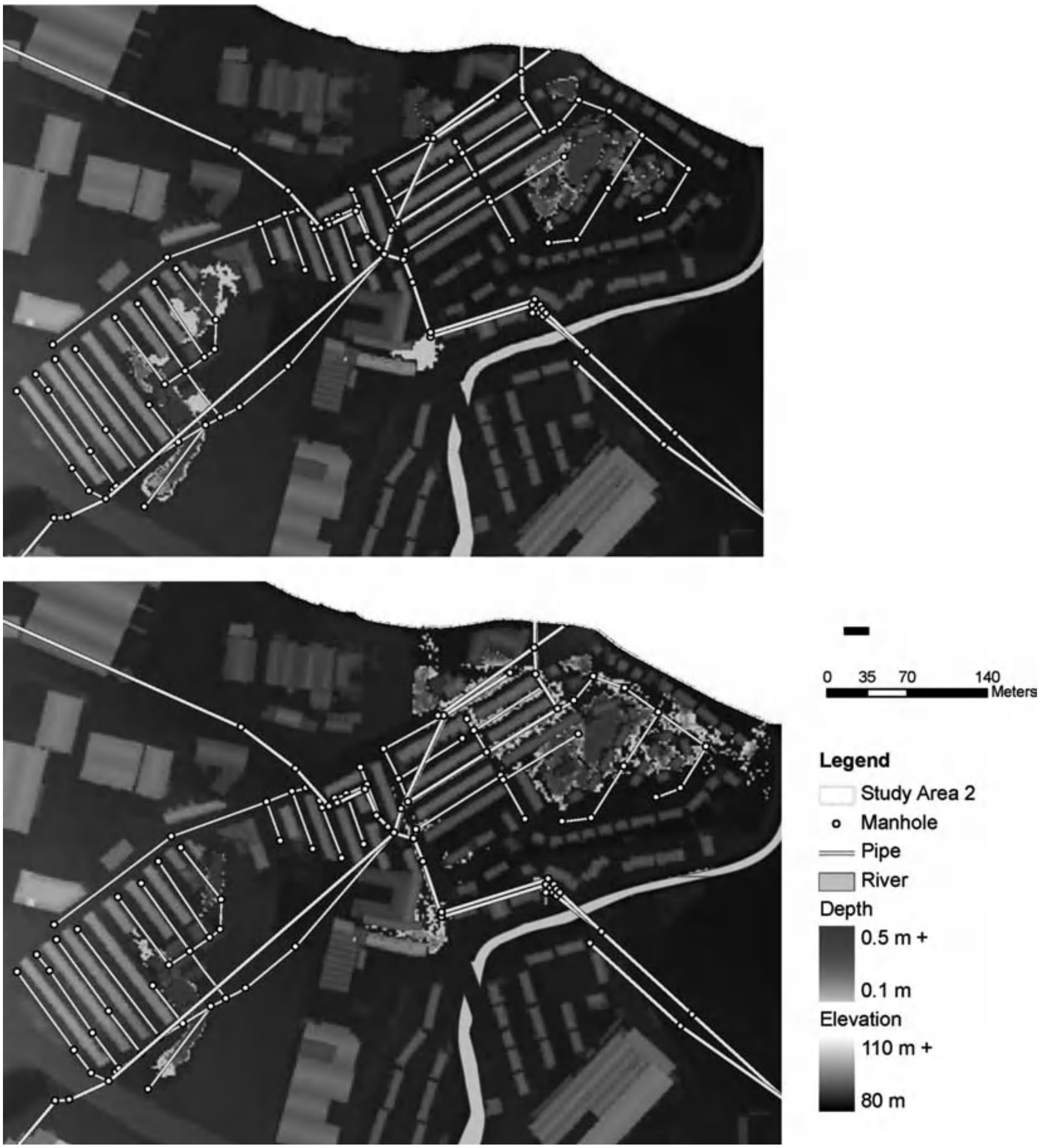

Fig. 13.20 Simulation results of 1D/1D (above) and 1D/2D (below) modelling. (See colour version of this figure in Colour Plate section)

which urban flood risk is managed, and to assimilate the findings of the new research into everyday practice, by the provision and funding of three demonstration projects. The projects were led by industry consultants with the support of an industry-led steering committee and with the support of the research teams. The aims of the demonstration projects were:

- to apply and evaluate the new models developed in FRMRC RPA6, at sites selected by the project Steering Group, to support the better management of urban flooding;

• to recommend and incorporate any required improvements to the research models prior to their implementation in practice;
• to apply the new modelling techniques within a typical working environment;
• to test the applicability, benefits and limitations of the models;
• to report back to the UK water industry and model developers.

These aims and objectives were achieved through a structured programme of three demonstration projects. The complexity of the modelling process was increased on a project-by-project basis.

Selection of demonstration sites: data requirements

Details of the data requirements for consideration at the time of site selection are detailed in Table 13.1. The list is not exclusive but serves as a guide for any future urban flood study.

Catchments used in the study

Two catchments were used in the demonstration projects:

Demonstration Project 1: Cowes, Isle of Wight – contractor Richard Allitt Associates: Application of the 1D surface pathways model, developed by Imperial College, to the catchment at Cowes on the Isle of Wight: a small town with known and well-defined surface flooding problems

Demonstration Project 2: Torquay in Devon – contractor Torbay City Council: Torquay had well-documented large-scale flooding problems at the centre of a large urban area with some coastal interaction. The drainage area had been upgraded in line with the first edition of the Sewerage Rehabilitation Manual, and was recently used as one of the Making Space for Water Integrated Urban Drainage pilot studies. Hence, there was a good understanding of the system performance against which to test the new 1D FRMRC software. As with Cowes, the areas of large-scale flooding were well defined.

Demonstration Project 3: Cowes, Isle of Wight – contractor Richard Allitt Associates: This catchment was again selected for the assessment of the integrated 1D/2D minor/major UIM modelling techniques, developed at the University of Exeter. Here it was possible to make a direct comparison with the outputs of the 1D/1D modelling exercise, but also to make a comparison with the outputs from the 1D/2D water industry proprietary software Infoworks CS 2D. This latter model only became commercially available at a time when the FRMRC research was nearing completion.

Demonstration projects 1 and 2: Cowes and Torbay

The overall aim of these studies was for industry contractors to utilize the research software and to trial the application of the 1D/1D model. The scope of the projects was set out in the form of a series of steps related to the collection of data, model build, model simulation, model testing and reporting. Specific steps included:
• collect and assess all data – produce DEM, verified sewer model, etc.;
• review of catchment and incident history;
• identify discrepancies between DEM and sewer model;
• produce 1D surface model;
• surface pathways with existing modelling of contributing areas;
• surface pathways with reallocation of contributing areas;
• link surface and subsurface models;
• field surveys to identify points of connection and contributions from paved and roof areas (if necessary);
• develop a risk matrix and rainfall scenarios for the study;
• simulations;
• benchmark modelling of current system;
• integrated 1D/1D modelling of new pathways;
• integrated 1D/1D modelling of new pathways and reallocated contributing areas;

Table 13.1 Essential and desirable data requirements for integrated assessment of flooding

Data type	Description	Purpose	Supplier
Essential			
Evidence of current flooding	Data collected from interviews with flood victims including details of: causes (sources, pathways) and impacts (pathways and receptors); nature, extent, depth, duration and cost of flooding; number of deaths; number and ages of occupants affected; injuries, illness and impacts on personal circumstances; losses to businesses and employees. The cost and benefit of detailed surveying of pathways and receptors (over and above DEM/DTM data) to be established.	Establish priorities Assessment of tangible and intangible costs of flooding Verify integrated drainage models.	WSP/LA/EA/ES/Ins
Permanent autographic raingauge data	Availability of data. Assessment of cost and benefit of obtaining additional autographic data	Identification of rainfall return periods resulting in flooding Validation of integrated drainage models	WSP/EA/MO
Sewer system data review	Assessment of total length of sewer systems Assessment of percentage availability of essential data (function, shape, size, material, cover levels and depths) for sewer system	General description of sewerage systems within catchment	WSP
Sewer system model review	Assessment of existence and quality of sewer system model including level of detail around current problems and identified future developments Assessment of percentage availability of essential data (function, shape, size, material, cover levels and depths for modelled network) Assessment of percentage availability of data on structural and service condition in modelled network Assessment of cost and benefit of collecting additional data	Assessment of availability and fitness for purpose of sewer system model Assessment of potential for and cost of enhancement of sewer system models as basis for integrated minor/major system model	WSP
DTM/DEM data review	Availability and type of DTM/DEM data. Assessment of cost and benefit of collecting additional data	Modelling of surface pathways and receptors. Defining pitched and flat roofs for runoff model Defining connected areas for the runoff model	WSP/LA/EA

Watercourse system data review	Assessment of total length of watercourse systems Assessment of percentage availability of essential data for culverted and open watercourses (type, shape, size, material/soil type, cover/ground levels and depths for watercourse system and core network)	General description of watercourse systems within catchment	LA/EA
Watercourse system model review	Assessment of existence and quality of watercourse system models Assessment of percentage availability of essential data for culverted and open watercourses (type, shape, size, material/soil type, cover/ground levels and depths) for watercourse system models Assessment of availability of data on structural and service condition Assessment of cost and benefit of collecting additional data	Assessment of availability and fitness for purpose of watercourse system models Assessment of potential for and cost of inclusion of watercourse system models within integrated minor/major system model	LA/EA
Asset performance review	Assessment of deterioration in the performance of components of the major and minor drainage system, such as culverts and sewers due to blockage, deformation, collapse and sedimentation within pipes and the deterioration of ancillary performance	Establish impacts of asset deterioration on performance of minor and major drainage systems and identify capacity building and adaptation requirements	WSP/LA/EA
Map background	Availability and type of map background. Assessment of cost and benefit of obtaining additional map background data	Identification of impervious surfaces Identification of buildings and other features affecting natural surface flow pathways	WSP/LA/EA
Desirable			
SUDS system data review	Assessment of total length of SUDS systems Assessment of percentage availability of essential data (type, shape, size, soil type, ground levels and depths) for SUDS systems	General description of SUDS systems within catchment	WSP/LA
SUDS system model review	Assessment of existence and quality of SUDS system models Assessment of percentage availability of essential data (type, shape, size, soil type, ground levels and depths) for SUDS system models Assessment of availability of data on structural and service condition Assessment of cost and benefit of collecting additional data	Assessment of availability and fitness for purpose of SUDS system models Assessment of potential for and cost of inclusion of SUDS system models within integrated minor/major system model	WSP/LA

(*continued*)

Table 13.1 *(Continued)*

Data type	Description	Purpose	Supplier
Soil type review	Availability of data on soil types. Assessment of cost and benefit of collecting additional data	Determination of runoff Assessment of infiltration potential	CEH/BGS/Cranfield University
Geological data review	Availability of data on geological formations beneath and around the study catchment including the presence of permeable and impermeable strata, their dip and strike and locations of outcrops	Identify the potential for infiltration in the catchment	BGS/LA
	Assessment of cost and benefit of collecting additional data	Identification of the potential down-hill impacts of infiltration systems in hilly and undulating catchments	
Ground slope review	Availability of data on ground slopes. Assessment of cost and benefit of collecting additional data	Assessment of potential for SUDS systems	
Groundwater review	Review of presence of perched and deep groundwater tables	Assessment of potential for SUDS systems	EA
Aerial photography review	Availability and epochs of aerial photography. Assessment of cost and benefit of collecting additional data	Defining urban morphology; impermeable surfaces, surface vegetation for permeable surfaces for runoff model Defining urban creep and its impact on current and future performance	LA
Short-term sewer flow survey data	Availability of data for validating enhanced drainage models. Assessment of cost and benefit of additional short-term flow survey data in sewers and watercourses	Validation of integrated drainage models	WSP
Roof drainage data	Availability of deposited plans and any calculations from building control Assessment of cost and benefit of obtaining additional data	Assessment of potential impacts of climate change on flood risks in buildings resulting from inadequate roof drainage Inputs to building drainage models Inputs to integrated drainage models where there is significant contribution from large buildings	LA
Building drainage data	Availability of deposited plans and any calculations from building control Assessment of cost and benefit of obtaining additional data	Assessment of potential impacts of climate change on flood risks in and around buildings resulting from inadequate building drainage Inputs to integrated drainage models where there is significant contribution from large buildings	LA

BGS, CEH, DEM, digital elevation model; DTM, digital terrain model; EA, Environment Agency; Ins, insurance companies; LA, local authority; MO, SUDS, WSP, Cranfield University,

- post-process modelling results to compare outputs and assess costs and benefits of alternative approaches;
- develop options;
- post-process options modelling results to compare outputs and assess costs and benefits of alternative approaches;
- compare the outputs of the techniques and compare the costs and benefits of the different approaches;
- to report the findings of the study.

Demonstration project 3: scope of project at Cowes

The objectives for this case study were twofold. Firstly to apply and evaluate the new 2D modelling module available in Infoworks in a known catchment that had already been modelled using a 1D-1D approach; and secondly to use the same catchment to evaluate and further develop the SIPSON/UIM model. The Infoworks modelling was expanded to consider and evaluate two different approaches to coupling the subsurface (1D) system and the surface (2D) pathways. These used a simple weir coupling arrangement and a more sophisticated head-discharge coupling arrangement. Details of the tasks completed at Cowes are listed in Table 13.2.

Results

Full details of the methodology and results of each individual study are presented in individual reports made by each contractor to UKWIR (UKWIR 2009a, 2009b, 2009c) with a summary report UKWIR (2009d). In this chapter, the outputs of the studies have been summarized, with a view to providing guidance on the application of the different types of models to improve practitioners' understanding of the requirements to accurately model urban flood risk.

Summary findings

The demonstration projects highlighted the need for careful preparation of digital elevation models prior to their use in surface modelling. The study was able to satisfy the objective to apply and

Table 13.2 Phase 3 tasks

Infoworks 2D modelling (by Richard Allitt Associates)	SIPSON/UIM modelling (by University of Exeter)
	Outline of SIPSON/UIM model
Adapt the `Integrated' model from Phase 1	Import of sewer system data from Infoworks to 3DNet
	Import of surface runoff hydrographs and base flows generated by Infoworks
	DTM and 2D model resolutions
	Review of similarities and differences between Infoworks 2D and SIPSON/UIM data structures and simulation engines
	Initial and boundary conditions for all simulations
Pluvial modelling (without buildings)	Pluvial modelling without sewers (without buildings)
Pluvial modelling (with buildings)	Pluvial modelling without sewers (with buildings)
Pluvial modelling (with buildings and sewers)	Pluvial modelling (with buildings and sewers)
Site surveys	
Break lines and porous walls	
Creating the simulation mesh	
Model with 1D/2D coupling as weirs	1D/2D coupling scheme adopted in this study
Model with 1D/2D coupling as head-discharge relationships	Coupled 1D/2D modelling
Simulations	Simulations
Export of hydrographs	
Presentation of results	Presentation of results

evaluate the new 1D surface flow model developed by FRMRC.

Data collection and clean-up were a significant part of each study and the UKWIR contractors spent considerable time within the catchments to ensure that the elevation models were representative of reality. The base data for the three studies were 1-m horizontal resolution LiDAR with a vertical resolution of ± 150 mm. It is important that surface features that act as urban flood pathways, for example roads with kerbs and drop kerbs, should be accurately represented.

The use of LiDAR data to develop the DTM/DEM should be supported by other forms of data collection such as land and GPS surveys and site visits (walkovers). Specific issues are associated with flowpaths at bridges, embankments, walls and walls with gates, hedges and narrow gaps between buildings. Overall therefore the study reinforced the need for sufficient time and resources to be devoted to catchment familiarization and undertaking sufficient field observations so that all the necessary small-scale surface features that can divert or constrain flows can be incorporated into the model.

This study highlighted that within urban areas the definition of overland flow routes needs to be considered at both the 'macro' scale and also at the 'micro' scale. The vertical accuracy of the Digital Terrain Model was particularly important in this regard, and it was concluded that the vertical accuracy needs to be better than a typical 125-mm height kerb face.

The FRMRC overland flow software proved to be a relatively quick means of adding overland pathways to existing sewer models and, further to teething troubles with the data structure, the data could be quickly imported into the Infoworks CS software. It was recommended that the data clean-up operation be made automated.

The studies identified some key issues for urban drainage modellers. Flat areas create problems and the accurate identification of storage nodes and drainage pathways is paramount, especially where there are buildings across potential flow routes. In addition, in flatter areas some storage ponds lie within larger storage ponds and the interpretation of the nature of the drainage pathways and the definition of the appropriate weir outlets from storage ponds is complex.

The 1D modelling approach was demonstrated to be an effective approach for the modelling of relatively large surface areas.

The Cowes catchment was satisfactorily modelled using both the Infoworks CS 2D program and the SIPSON/UIM 1D/2D program. Both models were able to simulate the response corresponding to a historical flood event.

The 1D surface modelling approach should be seen as complementary to the 2D approach, which has been developed by several software houses since the inception of FRMRC. Although the 1D surface modelling approach is marginally more resource demanding at the data preparation stage, equivalent file sizes and simulation times are significantly less than for the 2D.

Two-dimensional modelling provides better graphical representation of the flooding process than the 1D model, but 1D is currently much more effective at modelling adaptive surface responses to flooding problems.

The 1D approach uses all the functionality of current software packages for water quality modelling. This means that it can be used to demonstrate the impact of adaptive responses on receiving water quality and also to quantify the impact of flooding from combined sewers on public health

Recommendations

The outputs from the three UKWIR-funded studies yielded several recommendations for improved urban flood risk modelling:

- Further research be undertaken to consider the benefits to be gained from using 1D/1D modelling and 1D/2D modelling in different parts of the same catchment. In theory the upper parts of catchments, where the flooding is predominantly 'conveyance' flooding, can be adequately modelled with a 1D/1D approach, whilst the lower parts of catchments or where there is flattening of the ground slope the flooding is predominantly 'ponding' flooding, which requires 1D/2D

Table 13.3 Comparison of the strengths and weaknesses of 1D and 2D surface modelling approaches

1D surface modelling		2D surface modelling	
Strengths	**Weaknesses**	**Strengths**	**Weaknesses**
	Currently needs external model building tool	Model build an integral part of current software	
	Model build requires additional time for operation of external tool and import of data to modelling software	Model build is a rapid process	
Uses same numeric outputs as for 1D subsurface model for a limited number of nodes and links resulting in simple post-processing	Relatively low quality graphical output	High-quality graphical outputs show depths, velocities, levels and movement of water over the surface	High-quality graphical outputs can misrepresent low-resolution inputs Large volumes of numerical output relating to each modelled surface cell make post-processing arduous
Simulation times relatively short and with minimal demand on computer resources enabling coverage of large areas			Simulation time relatively long, with considerable demand on computer resources, limiting the area that can be simulated, or the resolution of the surface model (Note 1)
Utilizes all the 1D modelling functionality of the host software, enabling simulation of the movement of pollutants over the surface during an event			Simulation of movement of pollutants over the surface not possible
The efficiency with respect to computer resources means that this approach can be used to prioritize areas for more detailed study and also simulate overland flows through areas that are not vulnerable to flooding	The relatively low quality of the graphical output means that this approach is not effective in informing non-technical stakeholders without significant post-processing	The quality of the graphical outputs mean that the outputs can be used to inform stakeholders of the existing problems and provide appropriate outputs for flood risk assessments	The demands on computer resources require pre-prioritization to identify areas where integrated modelling studies are required
The drainage system data can be quickly amended to represent new surface pathways and sinks in each option to be considered			The need to alter digital elevation models to represent different options is time consuming

modelling. Further research should be aimed at better defining the appropriate modelling approach for the different types of flooding and to give urban flood modellers better guidance on when to use 1D/1D modelling and when to use 1D/2D modelling.

- Further research is undertaken into the relevance of adequate inclusion of break lines and walls in urban flood modelling.
- Further research is undertaken into the use of Infoworks operating in a combined 1D only, 1D/1D and 1D/2D manner in different parts of a catchment to determine whether it could usefully be applied to real-time or near-real-time flood forecasting.
- Further research is undertaken into the use of 2D models to simulate pluvial runoff and infiltration through ground surfaces, and whether 2D modelling in urban areas can replace the traditional hydrological models used in urban drainage modelling.
- That further catchments, with different ground slopes, be modelled in both Infoworks and SIPSON/UIM to further explore the differences between irregular- and regular-grid 2D models, and the discrepancies in results obtained by different 1D/2D coupling approaches and parameters and in relation to the simulation of supercritical flows.
- That the outcome of the ongoing FRMRC2 research work, especially in relation to gulley performance, is made available for inclusion in future software updates and tested on a range of case studies.
- It is clear that urban flood modelling is in its infancy and that much further interesting research and model testing need to be completed.

Concluding remarks

A summary of the relative merits of 1D and 2D modelling approaches is shown in Table 13.3. Examination of this information highlights that 2D

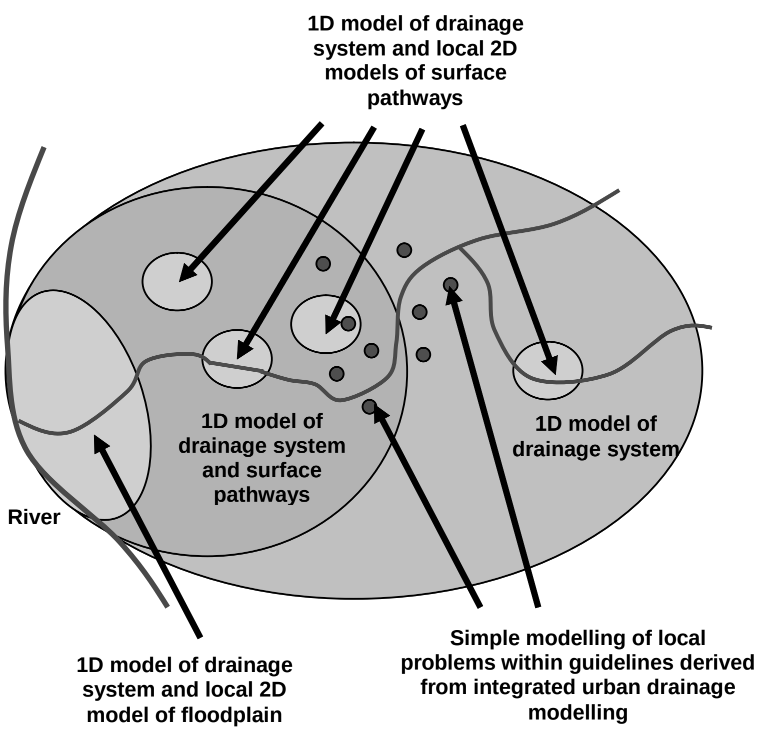

Fig. 13.21 Conceptualization of future integrated urban drainage and flood risk model.

overland flow modelling offers some benefits over the 1D approach, but that the 1D approach is effective in terms of simulation time and in simulating the performance of large urban areas. It is recommended therefore that the industry supports the development of a 1D surface modelling tool to quickly and cost-effectively identify areas of potentially high flood risk and for the assessment of options. The outputs from the 1D model will then provide the necessary boundary conditions to complete pockets of 2D modelling, as highlighted in the future conceptualized modelling framework for integrated urban drainage and flood risk management, as detailed in Figure 13.21.

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Part 5
Systems Modelling and Uncertainty Handling

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14 Distributed Models and Uncertainty in Flood Risk Management

KEITH BEVEN

The Requirement for Distributed Models in Flood Risk Management

Post-event analysis of any particular flood event will reveal that both the rainfall or snowmelt inputs that caused it and the effects in terms of areas flooded and damages caused will be spatially variable or distributed in nature. The hydrology and hydraulics of the event will reflect the heterogeneities in the driving variables and catchment and channel characteristics. The distributed nature of the processes is important, and the logical consequence is that in trying to predict flood events for flood management purposes we should use distributed models whenever local distributed inputs interact with local nonlinear processes to produce responses where the distributed impacts might be significant.

Early discussions of distributed models focused, generally rather optimistically, on such advantages of distributed models (e.g. Freeze and Harlan 1969; Freeze 1978; Beven and O'Connell 1982; Beven 1985). At the time these were potential future advantages. They included the possibility of directly measuring or estimating 'physically based' parameter values; the value of making use of distributed input conditions; the possibility of producing local predictions; and the need for distributed predictions of output fluxes for other purposes (prediction of water quality variables; prediction of sediment mobilization, transport and deposition; velocity fields for pollutant dispersion; and habitat evaluation) or fluxes that might feed back to affect the transport of water (such as gulleying and changes in channel cross-sections during floods). Similar arguments are still being made (e.g. Loague and VanderKwaak 2004). In fact, these advantages have been difficult to demonstrate, often for good reasons; see discussions by Refsgaard *et al.* (1996), Beven (1996a, 1996b), and the long history of trying to model the R5 catchment at Coshocton, Ohio, recorded in the series of papers by Keith Loague and colleagues (Loague 1990; Loague and Kyriakidis 1997; VanderKwaak and Loague 2001; Loague and VanderKwaak 2002; Loague *et al.*2005). Indeed, there is evidence that the application of distributed models may often fail acceptance criteria (e.g. Parkin *et al.* 1996; Choi and Beven 2007; but see Ebel and Loague 2006, for a declaration of success).

However, certainly in terms of both making use of distributed inputs and making distributed predictions where they are required in the catchment system, the use of distributed models provides the potential to reflect the spatial nonlinearities in the system more explicitly than any lumped approach. This is, however, at the cost of greater computational requirements (in that calculations must be made for every variable in every discrete element used in representing the flow domain), and the need to specify very large numbers of parameter values (in that all model parameters can potentially have different values in every discrete element).

The increase in computer power since the first distributed models were implemented on digital

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computers some 40 years ago has meant that the computational restriction has become less of an issue over time. More runs of distributed models with finer discretizations can now been made. It has not disappeared as an issue, however, since the parameter definition problem has not gone away (which might require many runs to be made in model calibration or uncertainty estimation), and computational times for fine-resolution models over large catchment domains may still be long compared with the lead times required in flood forecasting. Thus when using a distributed model for large-scale systems, resolution will generally still need to be compromised, such as the 5-km grid used in the European Union (EU) European Flood Alert System, which makes distributed predictions of runoff generation for the whole of Europe driven by rainfall forecasts from the European Centre for Medium-term Weather Forecasts up to 10 days ahead (see De Roo *et al.* 2003).

There is a further problem in the use of distributed models in forecasting, which is the number of state variables that could be updated in real-time data assimilation. Weather forecasting models also have very large numbers of state variables of course, and now use data assimilation as a matter of course. This is one reason for the improvements in forecast accuracy over the last two decades. In that case, however, there are also large numbers of observations to be assimilated. In the flood case, we may have observations of water levels at only a small number of sites. The information content of a forecast innovation (difference between observed and predicted values) may then not be sufficient to support assimilation of the large number of distributed model variables without making rather strong assumptions (see 'Data assimilation issues in using distributed models' below). Thus distributed models might still not be the best strategy for real-time forecasting problems (see Chapter 9 [Young], this volume).

In flood risk assessment, however, distributed models might be much more useful. The requirement is then to prioritize the local areas at greatest risk by coupled hydrological and hydraulic routing. The results will be dependent on the specification of the large number of inputs and parameter values needed to run the model (see 'Calibration issues in using distributed models' below) but only a distributed model can provide this type of local information. Under Section 105 of the Water Resources Act (1991) in the UK, the Environment Agency has been charged with providing flood risk maps for all areas at risk of flooding in England and Wales. It has done so by commissioning distributed inundation model predictions for all major flood plains, generally for the 100-year return period event (with and without an allowance for future climate change). This requirement is being extended by the EU Floods Directive, which states that all countries in the EU should produce such distributed flood maps by 2013, and catchment flood risk management plans by 2015.

It is therefore worth considering why the initial optimism about the future use of distributed models has not been borne out by more recent developments and applications. We have, after all, access to far more computer power than 40 years ago; we have access to far better topographic data than 40 years ago; we have access to geographic databases on soil and vegetation; we have access to the distributed information in remote-sensing images; we have strong drivers to use those distributed data sources to make local predictions of, for example, the impact of land use and management on flood runoff production and water quality. Legislation, such as the EU Water Framework and Flood Directives, effectively requires such predictions and both data and available computing power should allow us to be much better at distributed modelling.

A primary reason why the distributed modelling effort has not been more successful in hydrology and hydraulics is the result of uncertainty: uncertainties in the representation of hydrological processes (model structure); uncertainties and incommensurability in input data; uncertainties in estimating model parameters; and uncertainty and incommensurability in the observations with which model predictions are compared. We will return to these limitations, and their implications for distributed flood inundation prediction, after considering the range of distributed catchment models currently available.

The Evolution of Distributed Models in Hydrology and Hydraulics

Early distributed models

The first distributed models in hydrology and hydraulics belong to the pre-digital computer age. The idea that different areas in a catchment might produce different amounts of storm runoff, which would then need to be routed to a point of interest, goes back at least to Imbeaux (1892), who calculated snowmelt runoff on the Durance river in France based on different contributing areas at different time delays from the catchment outlet. This time-area histogram approach was later used by Ross (1921), Zoch (1934) and Clark (1945) in the USA, and Richards (1944) in the UK. Robert Horton (1938) also made use of a predictive model that allowed for different infiltration characteristics on different parts of a catchment (Horton 1938; Beven 2004). Within these subcatchment areas, however, predictions of runoff were lumped, so that these early attempts at distributed predictions might be called semi-distributed (while noting that many modern distributed models are also of this type).

The freeze and Harlan blueprint

More explicitly distributed predictions of hydrological processes at hillslope and catchment scales, and distributed hydraulic models, had to await the more widespread availability of digital computers. In particular the 'blueprint for a physically based digitally simulated hydrologic response model' of Freeze and Harlan (1969) set the scene for most of the distributed hydrological models that have been developed since. Freeze and Harlan (1969) laid out the continuum partial differential equations required: two- and three-dimensional Darcy–Richards equations in the subsurface, and one- and two-dimensional equations of the depth-averaged St-Venant equations for the surface, how they might be internally coupled, and how they might be coupled to additional snowmelt and evapotranspiration components. Early implementations of this type of distributed model included the seminal papers of Freeze (1972) and Stephenson and Freeze (1974) using finite difference solutions to the partial differential equations for one-dimensional (1D) surface and two-dimensional (2D) (vertical slice) subsurface flows at the hillslope scale. Beven (1977; see also 2001b) also later introduced variable width, vertical slice, finite element solutions to 2D subsurface flow for hillslopes linked to 1D surface flow routing.

The first fully distributed catchment model based on the Freeze and Harlan blueprint to achieve more widespread use was the Système Hydrologique Européen (SHE) model (Beven *et al.* 1980; Abbott *et al.*1986; Bathurst 1986). The original SHE model was implemented, primarily for computational reasons as 2D saturated zone and surface runoff components, solved on a square grid, linked by 1D unsaturated zone components. More recently two different versions of SHE (MIKE SHE at Danish Hydraulic Institute and SHETRAN at Newcastle University) have been implemented with fully 3D variably saturated subsurface components, which avoids the internal coupling problems between the unsaturated and saturated zone components. The rapid increase in available computer power has also allowed much finer spatial discretizations to be used (although this also, as in any numerical approximation to time-dependent partial differential equations, requires the use of shorter time steps). There have been published applications of SHE that use grids up to 4 km in large catchments (Jain *et al.*1992), which, while allowing the inputs and characteristics of the model to vary spatially, clearly limits the extent to which the sub-grid-scale variability in hydrological responses can be resolved (and also compromises the calculation of any fluxes based on gradients in the continuum representation).

More recently the Integrated Hydrological Model (InHM) has been used to predict the R5 catchment in Chickasha, Oklahoma, and the Coos Bay experimental hillslope in Oregon, with variable sized finite elements of the order of 10 m (VanderKwaak and Loague 2001; Loague *et al.* 2005; Ebel *et al.* 2008). InHM uses a finite element solution of the continuum equations, with the advantage that the grid of solution elements is easily made finer where more detail is required in the representation of gradients or simulation outputs.

The limitations of the Freeze and Harlan blueprint

Freeze and Harlan blueprint provides a theoretically consistent continuum differential equation approach to defining a distributed hydrological model. As such it has inherent attractions in providing a structure that goes from local continuum to catchment scales. There are, however, three essential limitations in applying these concepts. The first is the solution of the differential equations. Since the differential equations are nonlinear and subject to arbitrary changes in boundary conditions in both space and time, they cannot be solved analytically but we necessarily have to resort to approximate numerical solutions on a spatial discretization of the catchment. Except for very small catchments (e.g. Ebel and Loague 2006; Ebel *et al.* 2008), the discretization will be much larger than the 'point' scales at which the flux equations, such as the Darcy–Richards equation, hold. Thus, unless there is homogeneity at the sub-discretization scale, gradient terms and fluxes should not be expected to be well represented. The effects of heterogeneity in the subsurface gradients and hydraulic conductivity fields do not average linearly so that even if the Darcy–Richards equation applies at the point scale, the physics suggests we should be using a different equation at the element scale. Thus Beven (1989, 2002), for example, argued that this type of distributed model should be considered as lumped at the element scale.

This is the case even if we assume that any effects of sub-discretization heterogeneities can be allowed for by fitting an **effective** parameter value at the grid scale (Beven 1989). The result will not be a solution to the point scale equations where the flow domain is heterogeneous (this applies to both surface and subsurface fluxes). Binley *et al.* (1989b) suggested that for the case of purely Darcian subsurface flow in a heterogeneous domain, then the use of effective parameters might be a useful approximation (though the effective value of hydraulic conductivity might not easily be related to the distribution of point scale values). However, where the subsurface interacted with surface runoff then no consistent effective parameter values could be found.

There is a related issue as to whether the continuum equations described by Freeze and Harlan are an adequate description of the actual flow processes of surface and subsurface runoff in real catchments. Irregularities in flow pathways leading to depth variability in surface runoff; the effects of 3D channel geometry in streamflows; and preferential flows in the soil might all mean that the depth and velocity averaged St-Venant equations and Darcy–Richards equation might not be adequate representations of the actual flow processes (see, e.g., Beven and Germann, 1982; Beven 1989, 2001b, 2006a, 2010). Some attempts have been made to address these limitations, such as using dual porosity soil characteristics or two flow domains in Darcy–Richards (e.g. Brontstert and Plate 1997) or simulations of hypothetical hillslopes with preferential flow elements (e.g. Weiler and McDonnell 2007; Neiber and Sidle 2010; Klaus and Zehe 2010), but there is no current agreement on what type of formulation should be used for real hillslopes, or how the (effective) parameters of a new formulation should be identified in any application to a real catchment.

The second issue is to know what the effective parameter values might be for every element in the model representation of a catchment. Even if we assume that effective parameter values are a useful approximation, and even if we might have some information about the variability in point-scale soil and surface characteristics, then defining effective parameter values for every element in the catchment discretization will again require simplification and approximation. We cannot make measurements for every element in the discretization, and getting any information at all becomes much more difficult as the hydrologically active depth becomes deeper (especially if it includes fractured bedrock layers).

The third issue is knowing what the boundary conditions are for the catchment. To predict the pattern of hydrological fluxes we need to specify input fluxes for precipitation, output fluxes for evapotranspiration (which may depend on the internal state of the system as well as vegetation pattern), and we need to specify the initial internal states at the start of a simulation for every element in the catchment discretization. Binley *et al.* (1989a) showed that the effects of the initial conditions for simulations of this type can affect predicted fluxes for periods of months; while, in one of the earliest applications of a distributed model to a real hillslope, Stephenson and Freeze (1974) already recognized the difficulty of validating such models when the initial and boundary conditions were necessarily uncertain.

Representative Elementary Watershed

Reggiani *et al.* (1998, 1999, 2000) took this as the basis for a quite different distributed description of hillslope and catchment hydrology. They based their 'Representative Elementary Watershed' (REW) concept on the subdivision of a catchment into landscape units, each of which might then have different process domains. This was, in part, similar to earlier semi-distributed approaches based on hydrological response units (HRUs) (e.g. Kite and Kouwen 1992). There is also an analogy with the landscape 'tiles' used in representing the land surface hydrology in several modern climate models used at even larger scales. Reggiani *et al.*, however, took this concept a stage further by listing the mass, energy and momentum balance equations for each component of the REW. These equations apply, without numerical approximation, to any scale of REW from plot to hillslope to landscape tile. The difficulty that then arises, however, is that the balance equations involve multiple boundary fluxes for mass, energy and momentum. There is thus a 'closure' problem of how to define these fluxes at a particular scale (e.g. Reggiani and Rientjes 2005). What is clear is that the definitions will be scale and heterogeneity dependent, something that has been neglected in current implementations of the REW concepts. Such implementations have been described, for example, by Reggiani and Schellekens (2003), Zhang *et al.* (2006), Varado *et al.* (2006) and Lee *et al.* (2007) but have generally assumed scale independence in REWs treated as homogeneous. The problem of not having measurement techniques for fluxes at the scale at which we wish to make predictions means that developing better representations of the closure fluxes will be both difficult and uncertain, but this is what is required to make real progress in distributed modelling (see the more extended discussions in Beven 2002, 2006a). Until such progress is made, applications of the REW concepts will essentially be subject to much the same limitations as those for the Freeze and Harlan blueprint described above. Beven (2006a) has suggested that, while the REW concept is physically consistent at any scale of discretization, how to represent the boundary fluxes on the basis of sub-element-scale properties and patterns of storage remains the 'holy grail' of hydrological modeling.

Distributed flow routing and flood inundation models

Similar issues have arisen in the development of distributed hydraulic models since the first explicit solution scheme for the 1D St-Venant equations appeared in Stoker (1957). Once digital computers became more widely available there were many different finite difference implementations of the 1D equations (e.g. MIKE 11, ISIS, HEC-RAS) and research on better solution methods for these hyperbolic partial differential equations (e.g. Abbott and Minns 1998). As more and more powerful computers became available, 2D solutions were produced using finite difference, finite element and finite volume solution methodologies, and solutions of the full Navier–Stokes equations are now available for fully 3D flow domains, although these are still limited to small domains in any applications to rivers. New distributed techniques, such as Smoothed Particle Hydrodynamics, are also being explored in applications to surface water flows (e.g.

Monaghan 2005; Rodriguez-Paz and Bonet 2005; Liu and Liu 2006).

The computational requirements of full 2D and 3D solutions are still demanding, so there have also been attempts to link simpler 1D channel flow solutions to simpler representations of the distributed floodplain. Thus LISFLOOD (Bates and de Roo 2000; Hunter *et al.* 2006) and JFLOW (Bradbrook 2006) both use a diffusion wave simplification of the St-Venant equations over the floodplain, while the 1D MIKE 11 and ISIS models can both be used with floodplain embayment elements that treat parts of the floodplain just as a volume-filling problem. Run times for the simplified JFLOW have also been reduced significantly (by a factor of up to 100) by making use of the highly parallel structure on Graphics Processing Units developed for supporting computer games (Lamb *et al.* 2009). Current use of distributed models in hydraulics to support flood mitigation is explored in more detail in Chapter 12.

Here we will concentrate on calibration and data assimilation issues in relation to uncertainty estimation in distributed hydrological and hydraulic models. The accuracy of hydraulic models has been improved in recent years by the availability of much finer-resolution data for the geometry of the floodplain. However, distributed hydraulic models also are subject to issues of parameter definition. Normally, the most important parameters are the channel and floodplain roughness coefficients, but these can be allowed to vary for every element in the solution domain. Roughness is something that can be back-calculated, of course, from measurements of point velocity profiles. Within a cross-section, velocity profiles are normally averaged to allow the back-calculation of a roughness value for that cross-section. But such point values are not what might be required for the model to provide good predictions, for two reasons. The first is that the model predictions are made for elements (or groups of elements when roughness is assumed constant over larger numbers of elements) within which the geometry and boundary characteristics may be changing and within which internal eddying caused by secondary currents and lateral velocity gradients might lead to additional momentum loss. The second is that when roughness is back-calculated from velocity profiles, it is done so assuming that the flow is steady and uniform, but the model is then used to predict non-uniform flows in floods. [There is a third reason that is worth mentioning in passing. Nearly all hydraulic routing models based on the St-Venant equations make use of the Manning uniform flow equations in defining roughness and calculating momentum loss. In his original papers on equations for uniform flow (Manning 1891, 1895), Manning himself rejected that equation in favour of something more complicated but dimensionally more consistent.] This is a clear example of where **effective** values of the parameters, appropriate at the discretization element scale, are required to get good predictions for both hydrological and hydraulic models. In fact this concept also needs to be extended to effective values of the input and boundary condition variables to get good predictions. Calibration of these effective values is an important issue in the practical application of distributed models.

Simplified Distributed Models

These unavoidable uncertainties in model structure, parameter values and boundary conditions have meant that there has been plenty of scope for the use of simpler models that do not pretend to have such a theoretical basis as the Freeze and Harlan blueprint or the REW concepts. Some of these models have been around for some time and were developed when computational limitations were much more severe, but with the aim of making use of GIS and remote-sensing databases. Others have been aimed at large-scale (national or global) prediction systems, when large numbers of spatial elements might be required and computation time remains an issue. They may be subdivided into a number of broad categories in several ways: here we will distinguish models that neglect transfers between distributed elements (except in the channel network); models that treat such transfers implicitly or analytically; and models that treat such transfers explicitly. More discussion of this class of simplified distributed models may be found in Beven (2001b).

Within the first class are models based on 'hydrological response units' (HRUs), in which each response unit is treated as an essentially one-dimensional element from which any predicted surface runoff or subsurface drainage reaches the stream channel network directly. An example is the SLURP model of Kite and Kouwen (1992), which made use of soil and vegetation maps to divide the landscape into a number of functional classes, or HRUs; this approach has been perpetuated to the present day in a large number of 'land surface parameterizations' (LSPs) used as the lower boundary conditions of numerical weather predictors and general circulation models of the atmosphere. Recent examples of the latter are the NCAR Community Land Model (Bonan *et al.* 2002) and the Jules model in the UK (see http://www.jchmr.org/jules/science/science.html). Both rely on a similar strategy of functional class units to represent the hydrology within a global circulation model (GCM) calculation element, neglecting any exchanges between units.

This is an important problem with this type of model based on spatially distinct HRUs. In any landscape with soil-covered hillslopes, topography is important in controlling land surface hydrology in terms of both evapotranspiration and runoff generation processes. The lower parts of hillslopes tend to have more water available for evapotranspiration as a result of flows from upslope. Since they are generally wetter, they are also more likely to act as contributing areas for surface and subsurface runoff. Within any rainfall regime, these types of responses are structured by the topography and soil (and, in many catchments, the underlying geology). Thus it would also be useful to be able to reflect these controls in a simple way.

One way of doing so is to make use of analytical approximations to more complete mathematical representations of the flow processes, but in such a way that the results can be mapped back into space. One widely used example of such a model is Topmodel (e.g. Beven and Kirkby 1979; Beven 1997, 2001b). This makes use of a simple analytical approach in which the storage in the hillslope soils is assumed to be distributed as if it was in steady state with a distributed recharge equivalent to the subsurface discharge from the slope. Within this approach, the level of saturation deficit or water table in the soil at any point in the catchment can be related to a topographic index involving the area draining through that point from upslope and the slope angle at that point. A similar approach was developed independently by O'Loughlin (1986), while more recent variants taking account of the unsaturated zone and perched water tables have been suggested by Liu and Todini (2002; Topkapi) and Scanlon *et al.* (2000). The important feature of Topmodel is that the predictions of the model can not only be simplified by discretizing the distribution function of the topographic index in the catchment, and then mapped back into the catchment if the map of the topographic index is known. This results in a computationally efficient model that can still produce distributed predictions that can be evaluated for realism (e.g. Beven and Kirkby 1979; Seibert *et al.* 1997; Güntner *et al.* 1999; Blazkova *et al.* 2002) in addition to evaluating discharge predictions. The major limitation of Topmodel lies in its simplifying assumptions, which will only really be appropriate in humid catchments with relatively impermeable bedrock and moderate topography. They will certainly not be appropriate simplifications in many other catchments.

The final class of simplified distributed models includes those that take account of surface and subsurface transfers of water on hillslopes by explicit routing between the spatial elements of the catchment discretization. The discretization might be based on triangular irregular networks (Delauny discretization) or square grids. The former generally allow a better representation of hillslope topography (e.g. the model of Ivanov *et al.* 2004); the latter can more easily use the raster grid, which is used to store many types of data in geographical information systems, including remote-sensing data. Examples of grid-based models are the LISFLOOD model of De Roo *et al.* (2000); the ARCHydro model (Maidment 2002);

the DVSHM of Wigmosta *et al.* (1994); the Inter-Agency Object Modelling System (OMS) of Leavesley *et al.* (2002); and the Grid to Grid model of Bell *et al.* (2007; Cole and Moore 2008). A modification of this approach to avoid the computational burden of a large number of spatial elements is to route the runoff between hydrological response units. The dynamic version of Topmodel (Beven and Freer 2001b) is of this type, implementing explicit routing between elements to relax the steady-state assumption of the original Topmodel.

There is an interesting question about how accurate these types of simplified distributed models might be in reproducing the actual characteristics of hydrological processes. In general, they will have an advantage of computational efficiency over numerical solutions of the full-continuum partial differential equations of the Freeze and Harlan blueprint. However, as noted earlier, there are real issues about whether the Freeze and Harlan blueprint, despite its theoretical rigour, is a good description of the actual hydrological processes. There is also a real lack of distributed datasets with which to test the spatial predictions of any distributed model. Thus, it follows that it might be rather difficult to distinguish whether one form of distributed model is more 'realistic' than another (Beven 2008). We will return to this issue below (see 'Prediction uncertainty in distributed models').

Calibration Issues in Using Distributed Models

In the previous section we highlighted the fact that distributed models in both hydrology and hydraulics require very large numbers of parameter values and that any model will require **effective** values of those parameters to provide good simulations. In principle a distributed model can use different values of the parameters for every element in the discretization. Since there may be thousands of elements, there can be many thousands of parameter values required. With lumped models involving only a small number of parameters it is normal practice to calibrate a model by changing parameter values until a good (or at least acceptable) fit is obtained between observed and predicted values. Clearly this is much more difficult to do with a distributed model. We might suspect that the model results will be more sensitive to some parameters than others, but even a small number of sensitive parameters can be varied in many different ways to achieve the same type of behaviour. In particular, in distributed models, raising the value of a parameter in one part of the domain might be compensated by decreasing the value of that same parameter in another part of the domain. Once interactions with other parameters start to be considered, then there will be many many different combinations of parameters that might give similarly acceptable predictions (Beven 2006b).

There are three obvious ways around this problem. The first is to measure the parameter values at the sites where they are needed. This approach has significant limitations: partly because of the expense that would be involved in measuring parameters everywhere; partly because subsurface parameters cannot generally be measured by non-destructive methods; partly because most measurement techniques provide point measurements of parameters, which may be different from the effective values required by the model. The parameters may have the same name, but may not have the same meaning (they may not be commensurate).

The second way is to specify parameter values on the basis of the physical characteristics of an element, as related to past experience in applying the modelling concepts. This might be soil texture in specifying soil hydraulic characteristics; it might be vegetation cover in specifying evapotranspiration parameters; it might be vegetation density in specifying the surface roughness of a floodplain. There are existing databases that facilitate this process such as the pedo-transfer functions for estimating soil hydraulic conductivities; for example, the US Department of Agriculture (USDA) Rosetta system (see http://www.ars.usda.gov/Services/docs.htm?docid=8953; Schaap *et al.* 1998); the US Geological Survey (USGS) website (http://wwwrcamnl.wr.usgs.gov/sws/fieldmethods/Indirects/nvalues/

index.htm) for estimating channel roughness on the basis of similarity with photographs of a wide variety of channel types; or the UK Conveyance Estimation System (http://www.river-conveyance.net/) (CES) for channel roughness estimates based on field studies and large-scale laboratory experiments in the Wallingford Flood Channel Facility (Knight and Sellin 2007). For other parameters, a literature search might reveal values that have been measured or used in models in 'similar' flow domains. This approach also has its limitations in that many of these types of relationships are based on the back-calculation of parameters from point scale (or single cross-section scale) observations, not at the scale at which **effective** parameter values are required by a model. Again, the parameters may have the same name but may not be commensurate.

The third approach is to reduce the number of parameter values to be calibrated by assuming that the effective parameter values can be homogeneous over all or large parts of the flow domain and then calibrating by comparing observed and predicted variables. It is common, for example, in applying hydraulic models of overbank flows to assume that the channel can be characterized everywhere by one roughness coefficient and the floodplain by a second roughness coefficient. This dramatically reduces the number of parameters to be calibrated (although there may still be a significant number in a distributed rainfall-runoff model, even if they are assumed to be homogeneous over a catchment area). It also produces effective values of the parameters, since the values obtained by calibration will be those that produce good results in simulating the calibration data.

However, there are two important consequences of this third approach. By assuming that effective parameter values are homogeneous across the flow domain, it means that the predictions might be in error everywhere since they cannot properly reflect the local heterogeneities and uniqueness of the catchment or channel and floodplain characteristics (e.g. Beven 2000). Those errors may not then be random, creating difficulties in applying statistical parameter inference techniques. Secondly, the calibration process will necessarily reflect any errors in the specification of the model inputs and boundary conditions. Since these cannot be known precisely (or, very often, accurately) the calibrated parameters will be conditional not only on the implementation of the model equations but also on the specification of the boundary conditions. This was recognized early on in the history of this type of distributed modelling by Stephenson and Freeze (1974). In their distributed hillslope hydrology model, they understood and discussed that it would not be possible to fully validate this type of model prediction because the simulation results would depend on imperfect knowledge of initial and boundary conditions.

Depending on the model used, a final limitation of this type of distributed model calibration will be the limitation of available computing resources to carry out an adequate search of the parameter space in finding one or more acceptable models. When a distributed model takes a long time to perform a single run, then such a search can be unfeasible, especially where there are still significant numbers of parameter dimensions to be searched. An example is the study of the MIKE SHE rainfall-runoff model in Vazquez *et al.* (2008), where a calibration and uncertainty estimation exercise using this commercially protected code on a relatively coarse discretization of the Gete catchment in Belgium involved 15,000 simulations, which on average ran at 25 per day on a single PC.

When evaluating such simulations there is also the issue of how to define a model that gives acceptable predictions relative to the calibration data available. For lumped models, with small numbers of parameters, this has traditionally involved searching the parameter space until the global optimum parameter set is found and then taking this as the model of the system. We now know that this may not be the best approach. The optimum model found will not be the same if we use a different calibration period, or even a different realization of input uncertainty in the same calibration period, or if we use a different performance measure or combination of performance measures. Because of input and boundary condition error, the optimum model in calibration may also not give the best performance in validation (even if the model structure provides a good representation of the processes). We also know that for many commonly used performance measures there may be many different parameter sets that give nearly the same level of performance (Beven 1993; Beven and Freer 2001a). Thus, there are persuasive arguments that retaining only an 'optimum' model may be inadequate, even if the prediction uncertainty around that optimum is explored (Beven 2006b).

One of the early arguments for pursuing a strategy of distributed rather than lumped modelling was to avoid the problems that had been encountered in the calibration of lumped models. The argument was that if the model parameters were physically based and could be estimated by field measurement or on the basis of the physical characteristics of a catchment, then not only would we move towards a more realistic representation of the spatial pattern of hydrological processes but also we would avoid these difficult calibration issues. Hydrological science would become more realistic – see the discussion between Beven (1996a, 1996b) and Refsgaard *et al.* (1996). This has, in the end, been difficult to demonstrate, and even the applications of distributed models to small catchments have had to resort to some calibration.

One of the most interesting studies in this respect is the application of several generations of distributed models to the R5 catchment at Chickasha, Oklahoma, by Keith Loague and colleagues. This is not a large catchment (9.6 ha). It was originally modelled as an infiltration excess overland flow runoff generation system using the QPBRRM model, with parameter estimates based on 26 point infiltration experiments (Loague and Freeze 1985; Loague 1990, 1992). Later, this was extended to a further 247 measurements on a grid of 25 m with additional transects of 2 m and 5 m spacing, with geostatistical interpolation to 1 m and 959 hillslope planes in the runoff model (Loague and Kyriakidis 1997). The results using only the measured and interpolated infiltrated parameters based on this detailed information were worse than in the original (calibrated) model of Loague and Freeze (1985). Later, VanderKwaak and Loague (2001) and Loague and VanderKwaak (2004) applied the more complete 3D subsurface/2D surface finite element InHM model to the R5 site. The results were better but still suggested that improved results might be obtained with a better representation of the subsurface flow processes. In a final paper, Loague *et al.* (2005) extended the subsurface flow domain to deeper layers. They were able to report a further improvement but that their storm by storm simulations were still very sensitive to the specification of the initial conditions for each event, which are difficult to define in the subsurface: the same issue of model validation recognized by Stephenson and Freeze (1974) 30 years earlier.

There has been another interesting application of the InHM model to the Coos Bay site in Oregon. This is a small (860-m^2), steep, channel head hollow that was extensively studied by Bill Dietrich and others until it failed. Again, where possible the soil and sapprolite parameters used in the model were based on the extensive database of field measurements. There were not enough point measurements in this case to interpolate a field of soil parameters, so these were treated as homogeneous in each of three layers, even though individual measurements of hydraulic conductivity in the sapprolite ranged over four orders of magnitude. It was found that while the model could reproduce the discharge from the site reasonably well, it could not reproduce the internal piezometer observations (Ebel *et al.* 2008). It was suggested that one reason for this was that the piezometers were affected by flow through fractures in the underlying bedrock, the presence of which had been revealed by the slope failure and which were thought to have played an important role in the failure (Montgomery *et al.* 2002).

These case studies both demonstrate that even in small catchments with detailed experimental information available, there are limitations as to how far a distributed rainfall-runoff model can predict the internal responses of the hydrological processes. All models should be treated as hypotheses about how the system functions, since we expect that even the best process theory available might not represent fully the complexity of the real catchment (see also Beven 2001a, 2001b, 2006a, 2010). Rainfall-runoff models, of course, depend heavily on representations of subsurface processes that are difficult to study experimentally so it might be hoped that the problems might be less for distributed hydraulic models. Unfortunately, it seems that this might not be the case. A number of studies that have attempted to calibrate hydraulic models against historical inundation data have found that while it is not too difficult to reproduce observations of water levels during a flood event at a gauging site, it seems to be very difficult to predict patterns of inundation correctly in the flow domain. In some cases, this might be because the observations of inundation are in error (e.g. Pappenberger *et al.* 2005a), but even allowing for uncertainty in such observations it has proven difficult to find sets of effective roughness coefficients that give good predictions of the patterns of inundation everywhere on the flood plain (e.g. Romanowicz and Beven 1998, 2003; Pappenberger *et al.* 2007a, 2007b).

Data Assimilation Issues in Using Distributed Models

Increasingly, spatially distributed input data (from telemetering raingauges, radar or quantitative precipitation forecasting) are available for use in flood forecasting and flood warning (e.g. Pappenberger *et al.* 2005b; Collier 2007). The European Flood Alert System, for example, takes ensemble predictions of precipitation inputs across Europe provided by the European Centre for Medium-range Weather Forecasting (ECMWF) up to 10 days ahead, and feeds these into the LISFLOOD FF model, run at 5-km grid scale, to provide flood alerts for all the major rivers in Europe (e.g. De Roo *et al.* 2003; Gouweleeuw *et al.* 2005; Thielen *et al.* 2009). The agencies responsible for individual basins might then run their own flood forecasting systems at shorter timescales.

Clearly, over a large catchment such as the Rhine, precipitation inputs in any flood event can vary significantly across the basin. It is therefore important to take account of the spatial distribution of inputs in flood forecasting. There is an important issue, however, as to whether it is necessary to use a fully distributed hydrological model to do so, or whether a simpler model run at the subcatchment scale might be sufficient. This is the approach taken in many operational flood forecasting systems, including the National Weather Service in the USA (normally based on subcatchment implementations of the conceptual Sacramento hydrological model); SMHI in Sweden, and the FEWS system for the Rhine in The Netherlands (both based on subcatchment implementations of the conceptual HBV model); and the UK Environment Agency's National Flood Forecasting System, which uses a variety of different models in different catchments (Whitfield 2005; Werner and Whitfield 2007).

An important issue here is the utility of real-time adaptation in flood forecasting. Even with a well-calibrated model for a catchment or subcatchment, it should be expected that the predictions for the next big flood event will be (to a greater or lesser extent) in error, even if only because of uncertainties associated with the estimates of catchment rainfalls (and the uncertainties are likely to be worse for events with localized heavy rain, or rain-on-snow events). Thus, it might be possible to improve predictions of the timing and magnitude of a flood peak, by making use of data assimilation in real time (see, e.g., Young 2002). There are a variety of techniques for real-time data assimilation (see, e.g., Beven 2009) but the most important data available in flood forecasting will be the current water level measurements in the hydrometric network. These can be compared with model predicted water levels to see whether the model is over- or under-predicting and make a correction with updating every time a new measurement becomes available at a site.

There is, however, only a limited amount of information in a water level measurement at a single site. It would not therefore be sensible to try to use that information to attempt updating the very many state or flux predictions of a distributed rainfall-runoff or hydraulic model.

Two simpler approaches are then possible. The first is to update only a model of the prediction errors for the distributed model, using, for example, a simple gain adaptation or stochastic error model. The latter is used, for example, in the Norwegian flood forecasting system (see Skaugen *et al.* 2005). The second is not to worry about a distributed model at all for real-time forecasting but to use a much simpler model for each subcatchment. Since the model will be used purely for forecasting, with real-time updating, it is not even necessary to maintain a hydrological mass balance; Romanowicz *et al.* (2008), for example, have used rainfall-water level and water-level to water-level nonlinear transfer function models in forecasting (see also Chapter 9). A network of subcatchment transfer functions and reach transfer functions, with local updating at sites with real-time data availability, can be used to extend the lead time of forecasts in larger catchments. This has the advantage of not needing to use the rating curve to convert level to discharge, which necessarily introduces uncertainty into the modelling process, particularly for overbank flood discharges, when the rating curve may be poorly controlled or changing during an event. It is also often predictions of level that are required in forecasting. It is level rather than discharge that determines whether a flood embankment is overtopped.

Prediction Uncertainty in Distributed Models

It is clear from the previous discussion in this chapter that in any application of a distributed model there are likely to be many sources of uncertainty that might impact on model predictions. There are uncertainties in model representations of the relevant processes; there are uncertainties in effective parameter values; there are uncertainties in the initial and boundary data required; and there are uncertainties in the observations used in evaluating model predictions. It would therefore be useful to evaluate the effects of these uncertainties on the model predictions. Because of the complexities and nonlinearities of distributed hydrological models, however, it is not possible to propagate these uncertainties analytically. One way of approximating the calculations required to estimate prediction uncertainties is to use Monte Carlo simulation. However, this creates a computational difficulty because of the high dimensionality of the potential uncertainties and the lack of knowledge of covariation between the different uncertainties. In principle, the spatial patterns of all parameters and the space-time error characteristics of the boundary conditions can all be uncertain. Thus, a very high number of Monte Carlo simulations would be required, even if we assume only a single model structure is feasible, and that the error characteristics of the parameter values and boundary conditions are known.

Computational issues

By definition, distributed models have a large number of computational elements and produce a large number of predicted variables. Thus, as with other distributed forecasting and simulation systems, such as atmospheric models, there is always a need to compromise between using the available computing power to reduce the size of the elements of the discretization and increasing the accuracy of the approximation, or to make more Monte Carlo realizations of the model. This perhaps would not be such an issue if we could be sure that feasible models were limited to a small region of the high-dimensional space of potential model parameter sets and boundary conditions, but Monte Carlo experiments have shown that this is often not the case. Given the various sources of uncertainty, models that give acceptable fits to any evaluation data may be scattered through the space of feasible models and boundary conditions. This is what Beven (2006b) calls the **equifinality** issue.[1] Equifinality has been

[1] In other disciplines non-identifiability and ambiguity are also used. Equifinality, the idea that many different conditions might lead to similar results, was introduced to indicate that this was a generic problem rather than a problem of identifying the best model, following Von Bertalanffy (1968; see also Beven 2009).

demonstrated for a range of distributed models from SHE (McMichael and Hope 2007; Vazquez *et al.* 2008) to Topmodel (e.g. Beven and Freer 2001a; Blazkova *et al.* 2002; Freer *et al.* 2004), groundwater models (Feyen *et al.* 2003) and hydraulic models (e.g. Aronica *et al.* 1998; Romanowicz and Beven 2003; Bates *et al.* 2004; Pappenberger *et al.* 2005a, 2007a, 2007b) within the Generalized Likelihood Uncertainty Estimation (GLUE) methodology (Beven and Binley 1992; Beven and Freer 2001a; Beven 2008). In GLUE, the concept of an optimum model is rejected. Models are treated as multiple working hypotheses about catchment functioning that can be rejected by different types of qualitative and quantitative model evaluation. Those that survive the evaluation (at least until new data become available) are used in prediction.

This, however, presupposes that an adequate number of acceptable models can be found by systematic, guided or crude Monte Carlo sampling in the space of potential models as hypotheses about how the system functions (while taking any useful prior information about parameters and boundary conditions into account). The potential for doing so will depend on how many potential sources of error need to be addressed, the computer time needed for a single model run, the available computer resource, and the shape of the likelihood surface in the model space. If there is a single, well-defined peak in the likelihood surface then it will be much easier to obtain an adequate sample of high-likelihood models than if the surface is a complex of multiple peaks and ridges of similar likelihood scattered through the high-dimensional model space. Experience suggests that the latter is often the case for even simple hydrological models. Indeed, there are good physical reasons for expecting this, especially in distributed models when, for example, there might be a trade-off between hydraulic conductivity or surface roughness values in different parts of the flow domain in producing similar output fluxes.

There are therefore good reasons to try and reduce the dimensionality of the space in this form of analysis to try to simplify the response surface. One way of doing so is to carry out a sensitivity analysis prior to any uncertainty estimation and then consider only the most sensitive parameters or boundary conditions in the analysis. There are different ways of doing sensitivity analysis (see, e.g., Saltelli *et al.* 2004), and what appears to be sensitive is often dependent on the form of analysis and measure of sensitivity used (e.g. Pappenberger *et al.* 2008). This is, at least in part, because local sensitivities and interactions between different parameters might differ in different parts of the model space.

Another technique useful for distributed models is to set a pattern of distributed parameters or boundary conditions *a priori* and then, rather than varying individual values in different places, to use a multiplier to change the values of the whole pattern at once. Sensitivity or uncertainty analyses are then carried out on the multiplier rather than on local values. While this may not always be appropriate where spatial information is available for use in model evaluation, it can clearly result in a drastic reduction in the dimensionality considered (and consequently in the computer time required to sample the model space). Even so, it might still be difficult to obtain sufficient samples to ensure an adequate sample of the likelihood surface.

Model likelihood issues

Clearly, the issues about sampling the likelihood surface raised in the previous section depend very much on the choice of how the likelihood of a particular model is to be evaluated. This issue has received a lot of attention in the recent hydrological modelling literature. In fact, there is an ongoing debate about whether equifinality of models is really an issue or whether it can be avoided by the use of formal likelihoods within the theory of Bayes statistics. Several authors have criticized the GLUE methodology because it has generally not used formal statistical likelihoods, even though GLUE is generalized in the sense that the user can choose to do so (as has been demonstrated, e.g., by Romanowicz *et al.* 1994, 1996). A formal likelihood is then just one choice of many forms of likelihood that could be used in model evaluation, a choice that may be justified when the strong assumptions required are justified.

At the centre of this debate is the issue of the information content of observations in informing decisions about which models are better (have higher likelihood) than others. Formal statistical likelihood measures effectively assume that the residual errors between models and observations can be treated as random variables, or can be transformed to a form in which they can be treated as simple random variables. When this is the case, then every residual can be treated as informative in the model evaluation or conditioning process. The longer the series of residuals, then the more constrained should be the models with high likelihood so that, given enough residuals, equifinality should not be a problem. This argument is made strongly, for example, by Mantovan and Todini (2006) but the example used is a hypothetical case where the model is known to be correct and the residuals are constructed so as to have simple form, and prior knowledge of that structure is used in formulating the correct likelihood function.

When hydrological models are applied to real catchment data the residuals are not expected to be of simple form and the model may not be identifiable from the data. The residual series are often highly structured, usually showing autocorrelation, heteroscedasticity (variance that changes with the magnitude of the prediction) and with non-stationary bias because of either model structural effects or non-stationary bias in the inputs. In this case, making simple statistical assumptions about the errors can lead to unrealistic conditioning of the likelihood surface (i.e. stretching the surface until only one small part of the model space has high likelihood). An example of this is provided by Thiemann *et al.* (2001). They assumed that the residuals were Gaussian, of constant variance and not autocorrelated – assumptions that were obviously not correct for their application (see discussion by Beven and Young 2003).

It is, of course, quite possible to define more realistic statistical likelihood functions, by taking account of spatial or temporal autocorrelation, or heteroscedastic variance, or a simple bias or model inadequacy function (e.g. Kennedy and O'Hagan 2001) and a proper approach to such a formulation should include the identification of an appropriate statistical model. Where the residuals have a non-Gaussian distribution it is also possible to use Box-Cox or Meta-Gaussian transforms so that the theory of Gaussian residuals can be used (e.g. Montanari 2005; Beven *et al.* 2008). This does not change the assumption that the final series of residuals is purely random and that every residual is informative. This may, in some cases, be a useful approximation for practical applications but in hydrological models, when epistemic uncertainties as well as random uncertainties may be significant, it will rarely be a correct approximation. Epistemic uncertainties are those associated with lack of knowledge. This term is sometimes used in respect of model structural error alone, but we should also expect epistemic (non-random) uncertainty in model inputs and boundary conditions. One result of epistemic uncertainty is that different periods of observation data (or, theoretically, different input error realizations) will lead to quite different models having apparently high likelihood (see Beven *et al.* 2008). Since epistemic uncertainty is generic in real rainfall-runoff modelling applications, we might wish to be more circumspect about over-conditioning in these (quite normal) circumstances.

Model rejection issues

A further issue that arises in this debate has to do with model rejection. When a hydrological or hydraulic model is calibrated using statistical likelihood methods, no model realization is ever rejected. The fitting is always carried out conditional on an assumption that the model is correct and that the (perhaps transformed) residuals can be treated as random. If the performance of a model is poor, relative to other models, in terms of the likelihood measure used, then that model will be given a low likelihood. The model with the highest relative likelihood, however, will survive even if its performance is actually poor (though the model as hypothesis might be reconsidered if the predictions do not appear to be presentable).

Effectively the error variance will expand to ensure that the uncertainty associated with the model predictions is sufficient to enclose (usually) the required fraction of the observations.

It is possible to compare the performance of different model structures in a statistical framework, for example by using Bayes ratios, but ultimately the user must make a subjective decision as to whether the performance of the 'best' model is actually good enough for the purpose of the particular application. In this aspect, current practice is not so different from traditional model optimization. The calibration process was always a way of being able to demonstrate some success in a hydrological model in the face of all the potential sources of uncertainty and error in both data and model structures (even if there was no real guarantee that the best model in calibration would perform equally well in prediction of future responses).

This lack of model rejection is perhaps unfortunate in terms of progressing hydrological science. It is one reason why Beven (2006b) suggested an alternative approach within the GLUE framework of setting limits of acceptability associated with each observation prior to making any model runs. The limits of acceptability might be based on what would be fit for purpose in an application or on a consideration of different sources of input and observation error. Models that fell consistently within the limits of acceptability would then be used in prediction (perhaps with some weighting based on past performance), those that did not would be rejected.

This idea of feasible models being treated as multiple working hypotheses about catchment functioning, to be rejected as new tests are made, has some resonances with the Popperian model of the scientific method, where the emphasis is on testing and falsification of hypotheses. The problem in applying such a principle in hydrological modelling is that we must be very careful not to make a Type II error of rejecting a good model because it does not give a good simulation of the available observations, when that may only be a result of driving the model with poor input data or comparing the outputs with observations that are not commensurate because of scale or other observational considerations (e.g. Beven 2010). It is usually only possible to test whether a combination of model, input and boundary condition data provide adequate simulations, and because the input and boundary condition data are processed nonlinearly by the model, it is very difficult to separate out the different sources of uncertainty (Beven 2006b, 2008, 2009; Beven *et al.* 2008).

It is perhaps a problem resulting from inadequate characterization of inputs and boundary conditions in hydrological models that in applying the limits of acceptability approach in practice it has been found that often **no** models produce predictions that are everywhere within reasonable limits of acceptability. Invoking a statistical analogy and allowing for some 'outlier' observations so that acceptability is relaxed to a condition of satisfying only 90% or 95% of the limits is a possible response to this. This can also, however, be problematic. The nature of hydrological records is such that the other 10% or 5% of time steps may actually be some of the most hydrologically interesting periods. An alternative is to relax the limits until at least some models become acceptable, and then decide whether the resulting predictions are fit for purpose.

Implications for distributed flood routing models

In flood management, the purpose of a distributed hydrological model is often to provide discharge predictions that can be used as the upstream and lateral boundary conditions for flood routing models. The flood routing models might then be used with design storms or historical data to map areas at risk of flooding, or they might be used in real-time forecasting to get a better spatial pattern of changing flood risk during an event.

In the past, the accuracy of hydraulic models has been limited by the availability of good floodplain geometry data. This has greatly improved with the more widespread synthetic aperture radar (SAR) and light detection and ranging (LiDAR) datasets, though there are still issues about how well floodplain infrastructure of embankments, field boundaries and other obstacles or pathways for the flow over a floodplain can be represented. The geometry of the channel itself is still a problem, since this requires relatively expensive field surveys (although more widespread use of scanning ultrasonic devices might also improve this situation in the future).

It has often been assumed, however, that since the physics of hydraulic models is well established, and there is considerable experience about what roughness parameters should be used, the major source of uncertainty is probably in the boundary conditions. This is the implicit assumption made when, for example, roughness coefficients are defined *a priori* and without uncertainty in flood risk mapping (in fact, often this is done without taking any account of uncertainty in the upstream input condition either). However, studies that have looked at the calibration of hydraulic model parameters by comparing predictions with observations of inundation extent have often found that it is difficult to obtain good reproduction of the observations everywhere and that rather wide ranges of roughness parameters can give somewhat similar predictive accuracy (see, e.g., Aronica *et al.* 1998; Bates *et al.* 2004; Pappenberger *et al.* 2005a, 2007a, 2007b).

This is the result of similar calibration problems to those of distributed hydrological models. In setting up distributed hydraulic models it is necessary to provide depth/roughness functions for every element (or for one-dimensional hydraulic models, in each reach) of the discretized flow domain. It is, in principle, possible to use different functions or roughness coefficients for every element or reach. This would, however, result in a very high dimensional parameter space, so to simplify model calibration it is often assumed that the channel can be represented by one roughness coefficient or function and the floodplains (or each type of surface on the floodplains) by another. This is a gross simplification, remembering that it is **effective** roughness values that are required to account for all forms of momentum loss within each element, but it reduces the dimensions of the calibration problem drastically. Consequently, however, we should expect that the solutions will not be accurate everywhere, and that there will be trade-offs in accuracy in different parts of the domain with different sets of parameters. Pappenberger *et al.* (2007a), calibrating a one-dimensional model of the River Alezette in Luxembourg in this way within the GLUE methodology, showed that none of the parameter sets tried could reproduce observed inundation extent for all cross-sections. They therefore suggested using parameter sets that were shown to give good results for important parts of the flow domain in making predictions for those places in the future. Different places, however, might require different sets of models as determined by matching the available observations from past events. New events could then be used to refine the choice of models (Beven 2007).

Towards 'Models of Everywhere'

The application of distributed hydrological and hydraulic models can be treated as a form of learning process about **places**. We should expect that models that up to now have seemed to provide acceptable predictions might not prove acceptable into the future. Management based on the predictions of such models should consequently be adaptive. Beven (2007) has presented this learning process in the context of 'models of everywhere' – effectively the idea that distributed models might be implemented for management purposes for whole catchments or whole countries, and that over time the accuracy in the local predictions might be increased and the uncertainty of those predictions might be reduced as either the sets of appropriate parameters or the appropriate model structures for different places are refined into the future.

Model calibration has always been used to reflect the local idiosyncrasies of places (catchments and river reaches) in the application of particular model structures, at least where data have been available to allow calibration. What is envisaged in the 'models of everywhere' concept is somewhat different. Once everywhere is represented in the system, and visualizations of model predictions can be accessed by stakeholders interested in particular catchments or river reaches, it is likely that the representations of different places (in terms of either effective parameter values or hypotheses) might evolve quite differently. In some cases (or for some purposes) it may be possible to simplify model structures without increasing prediction uncertainty, in other cases more complexity may seem to be supported. The concept is not a proposal for a single monolithic model structure applied everywhere; it is for a modelling platform that allows the representation of places to evolve, taking full account of modelling uncertainties.

Such an evolution requires, however, that the model predictions be tested locally. This could be a purely qualitative evaluation based on local stakeholder expertise (such as the involvement of stakeholders in the development of a flood model for Ryedale in Yorkshire by Oxford and Durham Universities). Where quantitative hydrological data are readily available, such as at established gauging stations, then this can be seen as a process of re-evaluation of model performance and/or recalibration over time. This will be, however, only a limited evaluation of the spatial predictions of a distributed model. The same problems of availability and commensurability of spatial data arise as in model calibration. It might, therefore, only be possible to implement such a learning process for the distributed predictions by making specific spatial surveys (such as incremental discharge measurement campaigns), using remote-sensing images (taking account of the uncertainty in interpretation from image to hydrological variables) or post-event evaluations, such as flood extent mapping after specific future events in evaluating the predictions of hydraulic models.

As noted above (see 'Model rejection issues'), we will learn most in this learning process when it is shown that the distributed predictions are inadequate (and local stakeholders will be very happy to point out obvious inadequacies). If the model predictions can be shown to be inadequate then it implies that some effort must be expended in finding out why. Model failure might be because of inadequate characterization of the input uncertainties, it might be because the model calibration is inadequate, or it might be because of structural inadequacies in the model (using the wrong hypotheses). The important point is that recognition of model failure implies that the representation of a catchment will be improved. If the model continues to produce predictions within acceptable (or unavoidable) limits of uncertainty then no improvements will be needed, even if it can never be guaranteed that a model is producing acceptable results for the right reasons (Kirchner 2006), suggesting that continued re-evaluation remains necessary.

Once such 'models of everywhere' are available, the learning process will result in place becoming as important as model structure. Some model structures might prove to be more appropriate in some places; elsewhere, other structures might be appropriate. Everywhere, the idiosyncrasies of particular places will mean that the effective parameters required to represent the processes in those places might vary significantly. The most important issue for the future of modelling in this context will then be how to inform the learning process. What types of data will it be most valuable to collect or make use of in deciding on appropriate model structures as hypotheses and effective parameter values? What new measurement techniques might be important in informing the learning process? How can uncertainty in the predictions for individual places and different scales of application best be constrained? In these questions lies the future of distributed modelling in hydrology and hydraulics.

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15 Towards the Next Generation of Risk-Based Asset Management Tools

PAUL B. SAYERS, MIKE J. WALLIS, JONATHAN D. SIMM, GREG BAXTER AND TONY ANDRYSZEWSKI

Introduction

Ensuring the acceptable performance of flood defence assets and the asset systems they compose is a considerable challenge. The wide variety in asset types (from natural channels to engineered walls, embankments, gates and pump systems) and the interaction between them and their physical setting further complicate the task. The concepts of system analysis, reliability and structured option searching all provide useful decision aids. These advanced tools and techniques enable critical assets and asset components to be identified and investment options to be compared and prioritized on a common footing (from data collection and further analysis through to actions to repair, renovate, replace or indeed remove assets).

Over recent years the principles, methods and tools to help support better asset management have significantly advanced (Environment Agency 2002 & 2004, 2010; Sayers and Meadowcroft 2005; Simm *et al*. 2006; USACE 1993 & 2008, HR Wallingford, 2008). All of these approaches recognize the need to prioritize limited resources to best effect (maximizing risk reduction and maximizing beneficial opportunities) whilst taking account of present and future uncertainties.

To provide meaningful evidential support the underlying analysis must be:

- **System-based:** Recognizing that the protection afforded to a given person, property or other valued feature in the floodplain (i.e. receptor) reflects the performance of the asset system as a whole and how it responds under a wide range of loads (and not the performance of an individual asset during a single design storm).
- **Evidence-based:** Recognizing the need for transparent and auditable/challengeable evidence, whilst formally acknowledging that much of this evidence is uncertain.
- **Hierarchical:** Allowing for progressive refinement of the data and analysis to reflect the demands of the decision at hand (being **just** sufficient to ensure a robust choice and one that further refinement would not alter).
- **Wide ranging:** Enabling fixed and operational defence assets to be seen as only one, albeit important, component of a wider flood risk management strategy (where structural and non-structural measures act in concert to manage flood risk, allowing the advantages of one action to compensate for disadvantages of another).

This chapter explores the state of the art in assessing the performance of individual assets and asset systems. It provides a discussion of reliability analysis and system-based risk analysis tools. It also provides a forward look towards the practical application of formal optimization tools that support the development of robust management strategies.

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Overview of Asset Management

An **asset** can be described as any feature that is actively managed to reduce the chance of flooding, including:

- a linear asset – e.g. a raised defence (levee or dyke);
- a point asset – e.g. a pump, gate or culvert trash screen;
- the watercourse – e.g. the vegetation and sediment within a channel;
- the coastline – e.g. a groyne, beach or backshore.

Delivering the whole-life asset management in practice as outlined in Figure 15.1, presents a number of analytical challenges, including:

- **Incomplete understanding of the asset base:** The physical dimensions and engineering properties of the asset base are often unknown or poorly resolved (although significant effort has been devoted to improving the data in recent years – see, e.g., Environment Agency 2007a; USACE 2008).
- **Incomplete understanding of structural/operational performance:** Assets are often a complex composite of structural components with spatially varying materials and profile. The physical processes that lead to failure are equally complex and often poorly understood and can be costly to analyse.
- **Variability of impact:** The potential impacts of failure can vary markedly in space, and hence not all assets are equally important nor require a common standard or condition.
- **Decision complexity:** The invariable complexity of an asset system and the floodplains they protect makes expert and engineering judgement difficult to apply. This often leaves asset managers with a rational doubt as to which action to take and when.

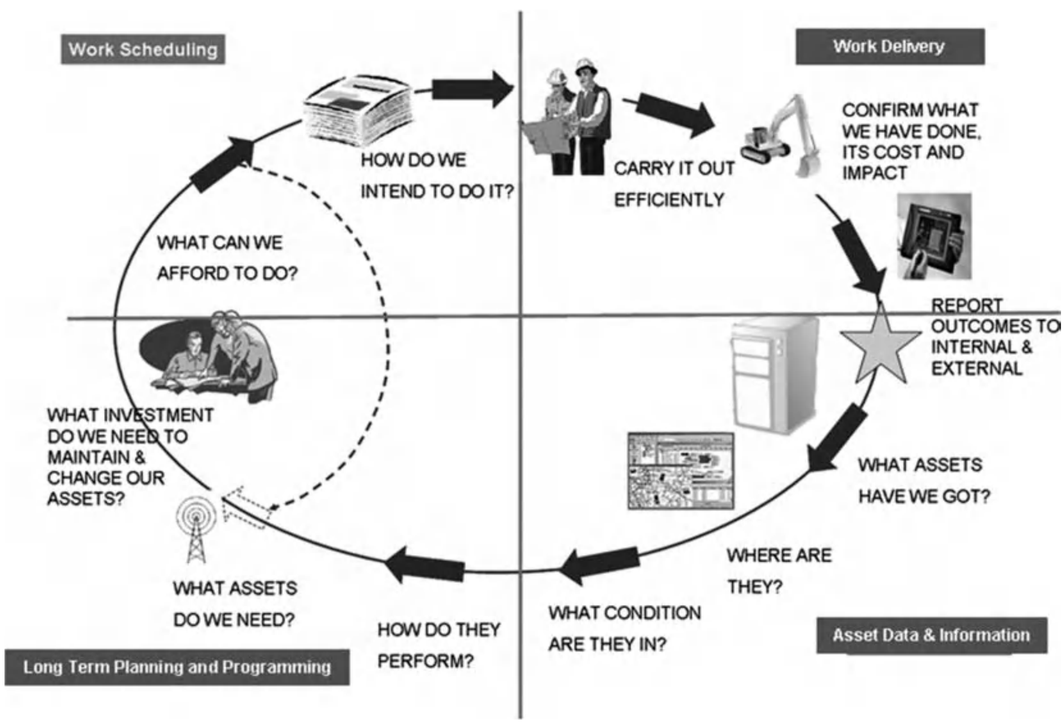


Fig. 15.1 Asset management lifecycle (Environment Agency 2010).

• **Affordability:** Budgets are limited and it is common to have insufficient resources (time and money) to undertake all 'desirable' works. For example, in the USA it has been estimated that $2.2 trillion would be needed to raise all linear defences (levees) to the desired standard and condition (Stockton 2009).

Better Asset Management: Rising to the Challenge

Around the world innovative tools and techniques are being developed to better support asset managers in overcoming the challenges they face (Havinga and Kok 2003; USACE 2008; Environment Agency 2010). This is in recognition that structural responses (including the ongoing management of existing assets) will continue to play a major role in flood risk management into the future (e.g. 'A Safe Public and Reduced Economic Losses by means of Reliable Levees – part of an Integrated Solution to Flooding', taken from the USACE, National Committee on Levee Safety, October 2008).

Common threads emerge from these activities, including a drive for better evidence on individual assets and better decision-making.

Better evidence on individual assets

In England and Wales, the Environment Agency has stated that it will have succeeded in its asset management role when it knows exactly 'what assets we have; where they are; what standard of protection they provide; how they were constructed; their current engineering integrity; and, how they work together to provide a flood defence system' (Tim Kersley – Head of Asset Management, Environment Agency, 2008). Similar, seemingly basic requirements, can be seen to exist around the world and across sectoral disciplines (within rail, road, etc.) and are a central thrust of the USACE National Levee Safety Program (USACE 2008).

Better evidence can be characterized as:

• *An improved understanding of the performance of the individual assets*– the importance of good-quality data cannot be overestimated, including direct access to basic parameters such as the location, condition and the standard of protection an asset affords as well as more accurate and usable information on probability of failure, dominant failure modes and critical uncertainties.

• *A better understanding of asset performance and the individual contributions of assets to the residual risk* – including direct access to information on how an individual asset contributes to the overall performance of the system and its contribution to residual risk (Fig. 15.2).

Better decision-making

All asset managers seek to make **good** investment decisions – decisions that minimize whole-life costs and maximize environmental gain whilst ensuring communities are appropriately protected from flooding now and in the future. Increasingly, consistent analysis techniques (Sayers and Meadowcroft 2005; Gouldby *et al*. 2008, 2009) and decision support tools (Surendran *et al*. 2008; McGahey and Sayers 2008; Environment Agency 2010) are available to support decision-making. These tools and techniques provide a step change in the 'richness' of the evidence provided to decision-makers at all levels (across national, flood system and individual asset levels). In particular, they provide:

• *An improved understanding of the role that an individual asset plays within a larger asset system* – by highlighting those assets, within a system of assets, that contribute most to residual risk. Further disaggregation of the risk, to highlight the separate contributions from breach and overtopping, enables the asset manger to distinguish the relative importance of raising crest heights or improving asset strength.

• *A better understanding of the impact of uncertainty within the estimated risk* – by highlighting those assets that contribute most to the uncertainty in the estimates of risk the asset manager is

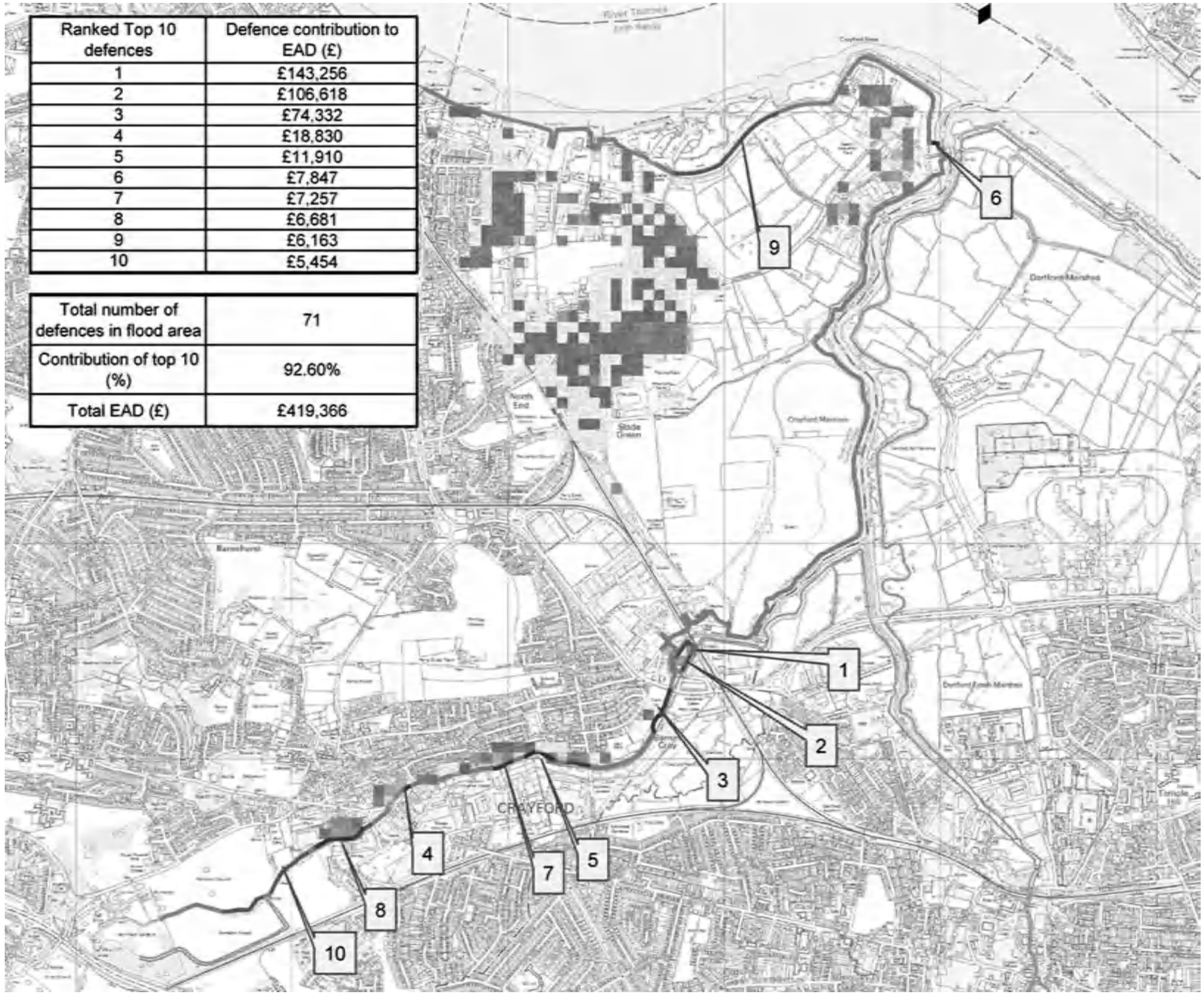


Fig. 15.2 The expected annual damage (EAD) attributed to individual defence assets within an asset system (as well as spatially within the floodplain). (See the colour version of the figure in Colour Plate section.)

able to prioritize the need for further data collection or engineering investigations on a common basis alongside structural measures. Structured sensitivity analysis (Gouldby *et al*. 2010) can help highlight critical epistemic uncertainties (such as within toe level, crest level, and asset condition as well as the modelling methods; Fig. 15.3). [**Note:** Model structure and local anomalies (reflecting the heterogeneity of the soil conditions for example) are not easily incorporated into such an analysis and continue to demand significant expert input.]

• *The ability progressively to refine the analysis detail* – attributing risk to an asset, and an associated understanding of the critical contributors to the uncertainty in that estimate, enables the decision-maker to target further analysis or data collection as appropriate for the decision in hand. Although a tiered analysis is a well-recognized concept (e.g. DETR 2000), until recently it has been very difficult to achieve a hierarchical process whereby data and models evolve (rather than change) from one tier to the next (Sayers and Meadowcroft 2005).

• *Support to develop optimal investment strategies* – asset managers face difficult choices: (i) **where** to act? (ii) **when** to act, now or later? and (iii) **how** to act, collect more data, undertake more analysis or intervene? Increasingly it is not possible, or acceptable, to answer these questions intuitively. The utility of formal optimization methods, and their applicability and practicality for use in flood risk management, is now being explored and trialled with considerable promise (McGahey and Sayers 2008; Philips. 2006; Woodward *et al*. 2010).

Asset Management Tools and Techniques: Key Features

Good asset management decision aids share a number of good practice principles (Table 15.1). The translation of these **good practice principles** into practical tools (which provide the richness of evidence described in earlier sections) is receiving considerable attention worldwide – e.g. Infrastructure Management Theme of the Flood Risk Management Research Consortium (FRMRC), the Environment Agency R&D programme and the US Army Corps of Engineers (USACE) research agenda. Typically these tools have a number of common features, as shown in Figure 15.4 and discussed below.

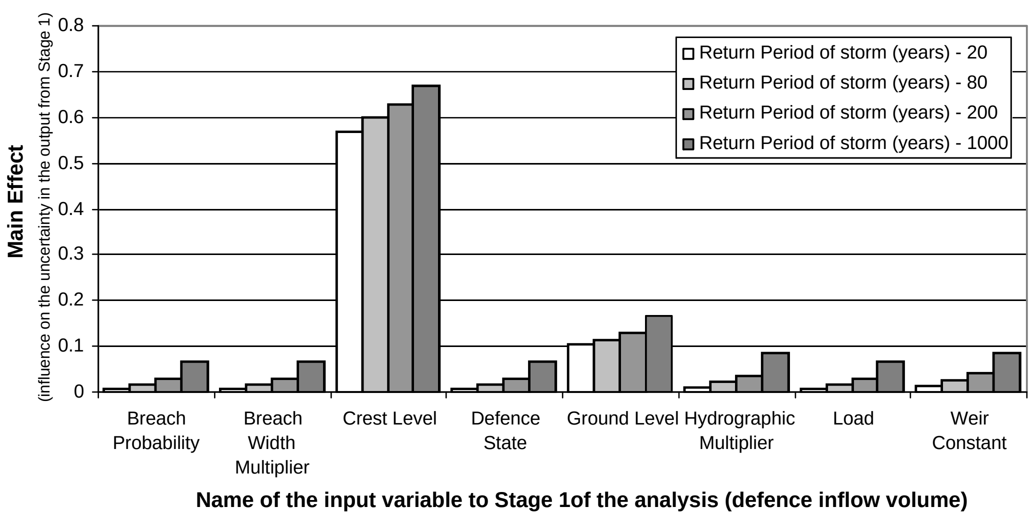


Fig. 15.3 Relative importance of the uncertainties within asset descriptors and model parameters to the uncertainty within the risk attributed to a specific asset.

Table 15.1 Best practice principles in support of asset management tools

Appropriateness	Appropriate level of data collection and analysis reflecting the level of risk associated with an asset and the uncertainty within the decision being made
Understanding	Improving understanding of assets and their likely performance
Transparency	Transparency of analysis enabling audit and justification
Structure	Structured knowledge capture encapsulated through fault tree, breach potential, etc.
Tiered assessment and decision-making	In terms of both data and modelling approaches
Collect once use many times	Reusing data through the hierarchy of decision-making stages and supporting tools – from national policy to local detail
Simple use and practical	There is a significant challenge in converting good science into practical tools. Therefore, even though the underlying analysis may be complex, the user experience must be well constructed and intuitive

Common/Central Databases

Common databases (Fig. 15.5) provide a means of accessing data and progressively evolving data quality (supporting a 'collect once, use many times' policy). The importance of such a system, and the difficulty in achieving it in practice across multiple stakeholders, should not be underestimated. Within England and Wales, for example, the National Flood and Coastal Defence Database (NFCDD) provides a common home for asset data – regardless of ownership – but significant difficulties associated with access and data quality have been encountered. Similarly, in the USA, a National Levee Database is currently under development. Although not without technical and organizational difficulties an NFCDD (or its equivalent) is a fundamental component of any asset management system, without which data collection and analysis activities are easily repeated and effort wasted.

Understanding the Performance of an Individual Asset

Understanding the performance of an individual asset under load is the first step towards understanding how best to manage it. The geometry and structural components of the asset together with the loading it experiences (e.g. waves, water levels, etc.) and the associated probability of failure are all important. Inspection methods (intrusive and non-intrusive; Long *et al.* 2008) and reliability analysis (see below) provide vital aids to the asset manager in developing this understanding.

Asset reliability

The reliability of an individual asset can be expressed in a number of ways. Typically two are used, including: (i) the probability of failure during a given time period (e.g. a year); and (ii) the conditional probability of failure for a given load (Fig. 15.6), referred to as a **fragility curve** (Casciati and Faravelli 1991; Sayers *et al.* 2002).

Reliability methods can be used to derive either the annual probability of failure or a fragility curve (Melchers 1999). This involves the specification and evaluation of a so-called Limit State Equation, in general form:

$$p(f) = P(R-S) < 0 \qquad (15.1)$$

where $p(f)$ = probability of failure (typically defined as breach, blockage or failure of a pump); S = loading on the asset; and R = resistance of the asset to the loads.

In traditional reliability analysis, the unconditional probability of failure (e.g. the annual probability of failure) is determined through integration of the joint density function (f_{RS}) of the loads and strengths over the region where the limit state is exceeded (i.e. $R - S < 0$):

$$p(f) = P(R-S) < 0 = \int\int_{af} f_{RS}(r, s) \qquad (15.2)$$

To derive a fragility curve, a similar process is followed. In this case, however, the **loads** are treated as known deterministic variables (hence

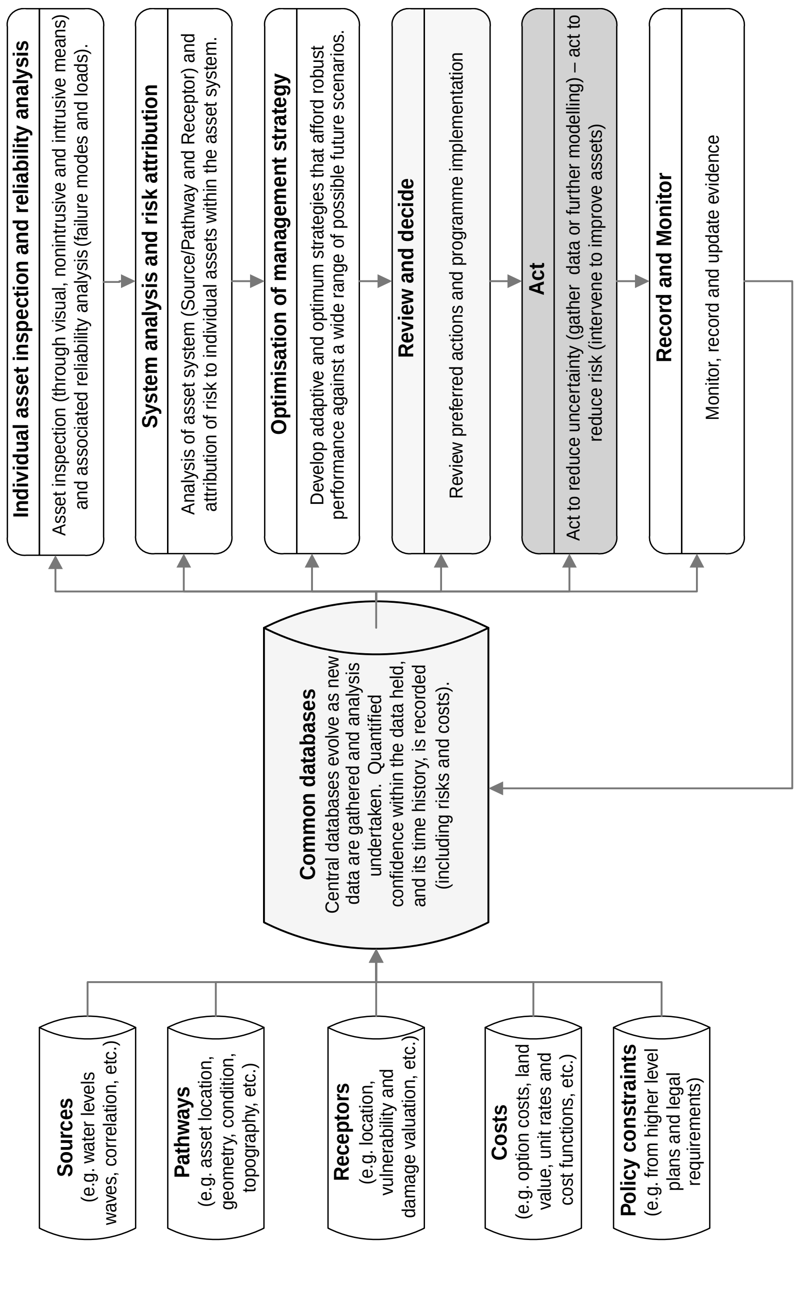


Fig. 15.4 Asset management tool-set – basic building blocks.

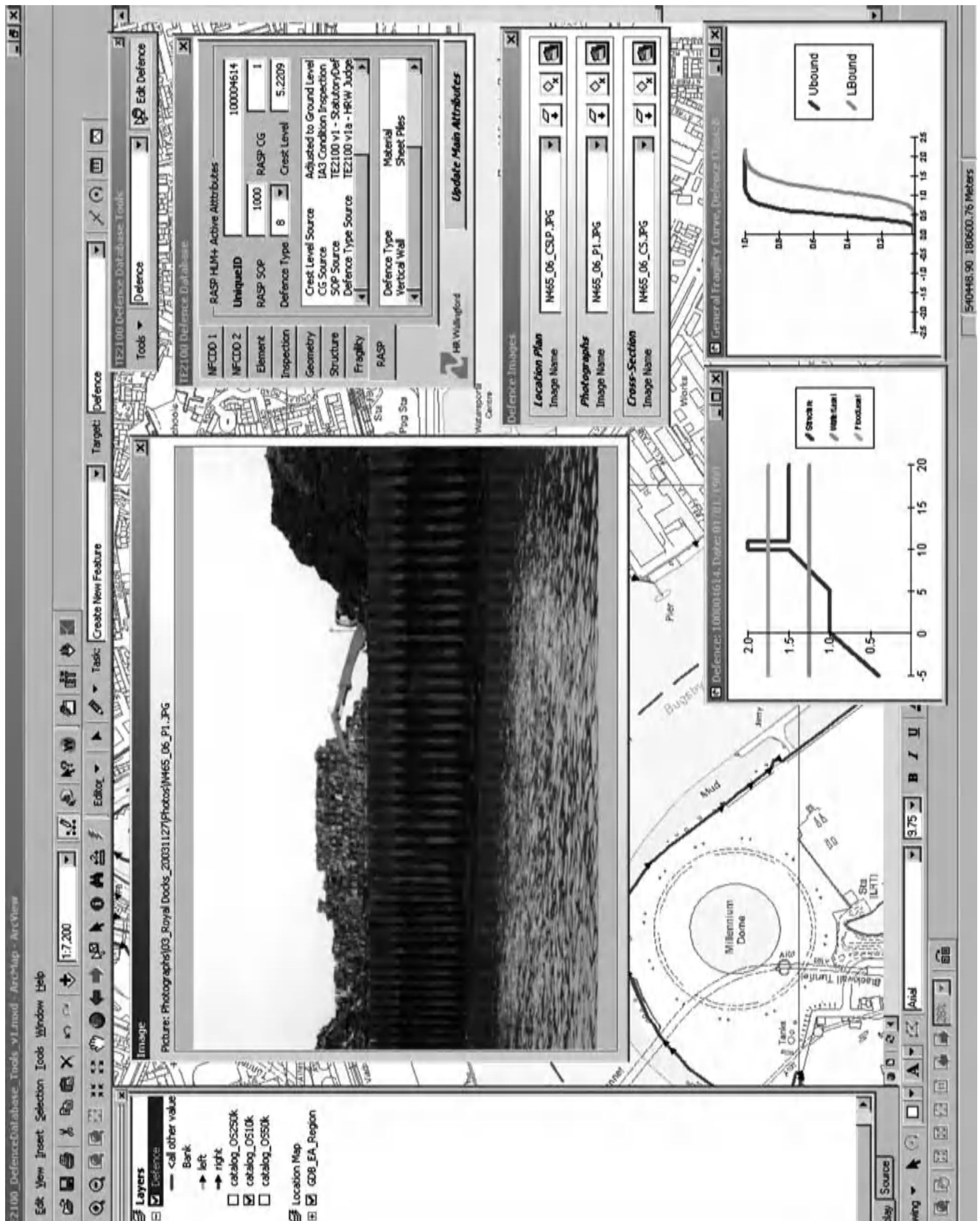


Fig. 15.5 A common asset database as used and shared across the Thames Estuary Planning for Flood Risk Management 2100 (Sayers *et al.* 2006). (See the colour version of the figure in Colour Plate section.)

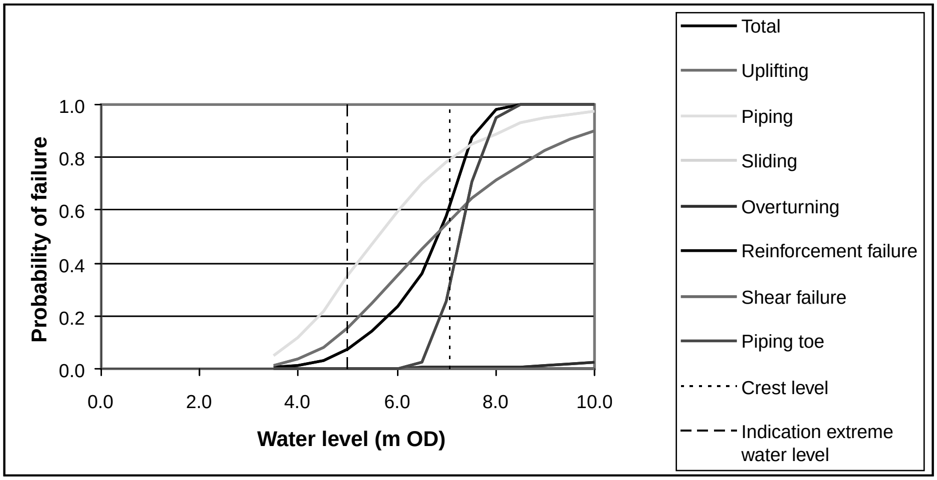


Fig. 15.6 A typical fragility curve based on the reliability analysis for a defence in the Thames Estuary. (See the colour version of the figure in Colour Plate section.)

the failure probabilities are conditional on the loads) and the probability of failure is assessed for specific loading events (by integrating the probability distributions assigned to variables and parameters that describe the **strengths** (S) of the asset over the failure region, i.e. the load exceeds the sampled strength).

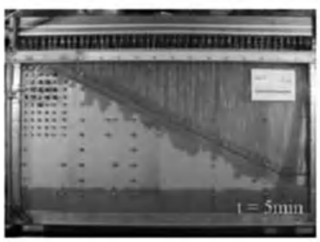
A set of high-level fragility curves that represent the typical assets found in the UK provide a common reference of asset fragility. These high-level curves are based upon a limited number of readily available asset characteristics (e.g. from the NFCDD). The high-level classification (as defined by the RASP (Risk Assessment for Strategic Planning) defence types; Hall *et al.* 2003) differentiates the assets first by seven major types (fluvial – not exposed to wave action; or coastal – exposed to wave action; vertical or sloping) and then by their width (narrow, $<$ 6-m crest; or wide, $>$ 6-m crest) and the nature and extent of the surface cover protection. A seventh classification of high ground is also included. A restricted set of limit state equations are then used within a reliability analysis to develop the fragility curves for each RASP defence based on three indicator failure modes:

1 Overtopping – periodic overflow of the defence due to wave action (coastal defences only).
2 Overflow – when the water level is above the defence (coastal and fluvial).
3 Piping – when the water level is below the crest level of the defence (fluvial only) (Environment Agency 2007a).

To determine an initial estimate of the fragility of a specific asset, the high-level fragility curves can be combined with local-scale data on asset condition (either measured or estimated), crest level of the asset, as well as the local loading conditions to which the asset is exposed. This allows the high-level fragility curves to utilize available local data (without increasing analysis effort). [**Note:** For example, Gouldby *et al.* (2010) provide a full list of the local parameters used to complement the high-level fragility curves as part of the National Flood Risk Assessment routinely undertaken for England and Wales.]

In many cases it is appropriate to refine the understanding of asset reliability beyond the high-level fragility curves described above. This may be in response to the importance of a particular asset in terms of managing risk (e.g. a major structure such as the Thames Barrier) or where doubt remains as how best to intervene and further investigation is required. A structured procedure to derive more credible asset-specific fragility curves is provided in Table 15.2.

Table 15.2 A structured procedure for the assessment of asset fragility (adapted from Simm *et al*. 2008)

Step	Description
1. Define asset function	A flood defence asset rarely acts solely to protect from flooding; often functioning as a valuable environmental habitat, navigation or amenity asset. Understanding the multi-functionality of the asset is an important precursor to understanding how to manage it
2. Establish incident loading	An asset may be subject to a range of loading conditions – joint wave and water levels, marginal high or low water levels, groundwater levels or perhaps a combination
3. Identify failure modes	The **failure mechanisms** (processes that can lead to ultimate failure) and the **failure modes** (a process that defines ultimate failure) also need to be described. To avoid unnecessary effort, conventional deterministic approaches can be helpful to eliminate unrealistic failure mechanisms (i.e. relative low-probability individual events in comparison with the likely overall reliability of the asset)  Wafe impacts on cracks (floodsite)  Surface erosion due to overtopping (floodsite)  Internal cracking (Floodsite)  Fine fissuring - FRMRC WP 4.1 Mark Dyer Research into failure mechanisms continues to be vital to better understand asset performance (e.g. Allsop *et al*. 2007; Dyer *et al*. 2009; Sentenac *et al*. 2009)
4. Prepare a fault tree	Fault trees provide a useful visual, and formal, encapsulation of the failure mechanisms and their relationship to the failure modes
5. Identify/establish appropriate Limit State Equations	An appropriate model needs to be selected to represent each failure mechanism\mode. In many cases empirical relationships will exist and these can be easily translated into the form of a Limit State Equation (utilized in the reliability analysis – see below). In some cases, the failure mechanisms are complex (e.g. slip failure) and demand the use of more sophisticated models (e.g. traditional slope stability analysis or finite element model). It is possible to link such models within the reliability analysis (Lassing *et al*. 2003; Vrouwenvelder 2001a, 2001b) but this is often difficult and can incur an unacceptable runtime overhead. Emulation of these more complex models, through Artificial Neural Networks, for example, provides an efficient and effective means to enable such complete mechanisms to be incorporated into the reliability analysis (Kingston and Gouldby 2007)
6. Document uncertainty in model variables and parameters	The engineering parameters, and the empirical variables, within the Limit State Equations will not be perfectly understood. Describing the uncertainty within these relationships and the supporting data on the asset of interest is an important task. In describing the uncertainty it is important that this process is comprehensive (ignoring uncertainty at this stage is to assume the data are perfectly known). Two groups of uncertainties can typically be distinguished (USACE 1999; Environment Agency 2002): • **Natural variability** (aleatory uncertainty) – Uncertainties that stem from known (or observable) populations and therefore represent randomness in samples

(*Continued*)

Table 15.2 *(Continued)*

Step	Description
	• **Knowledge uncertainty** (epistemic uncertainty) – Uncertainties that come from basic lack of knowledge of fundamental or measurable phenomena Perhaps most critically, it is important to record the assumptions made regarding the uncertainty in the variables and parameters and the associated supporting evidence for these choices. This provides a vehicle for peer review and audit (Hall and Solomatine 2008)
7. Undertake reliability analysis and display results	Once the above inputs have been established the reliability analyses can be undertaken. For each hydraulic loading condition a series of simulations (across the uncertainty bands for each input parameter) are resolved. Failure arises in a particular case when the combinations of parameter values in the limit state function (Z) yield a value for Z that is less than or equal to zero. The probability of failure for that given loading condition is then the number of times when the simulation gives Z as less than or equal to zero divided by the total number of simulations. Repeat for all hydraulic loads (Kortenhaus and Oumeraci 2002; Lassing *et al.* 2003; Simm *et al.* 2006; van Gelder *et al.* 2008)
8. Display results	Present the results of interest (annual probability of failure, fragility curve, etc.)

To support a detailed analysis of reliability, a flexible software tool (RELIABLE) has been developed based on a Level III reliability method. The basic building blocks of a Level III analysis and the way in which these have been enacted within RELIABLE are shown in Figure 15.7. In particular, RELIABLE contains:

- **A fault tree drawing tool (OpenFTA)** – enabling the user to construct an asset-specific fault tree using failure mechanisms (and modes) linked with standard operators (AND, OR, NOT, etc.).
- **A library of limit state equations** – limit state equations (LSEs) are made available to the user (currently describing 72 failure modes). These can

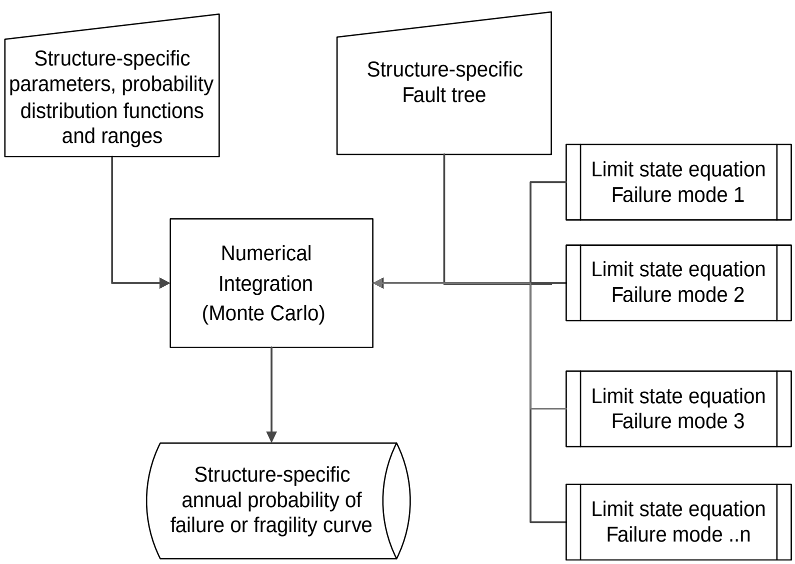


Fig. 15.7 Building blocks of a structured Level III reliability analysis (as implemented within the RELIABLE software).

be extended and made bespoke to specific assets through user-defined LSEs (either based upon empirical formulae not yet coded within the Failure Mode Library or based on the emulation of more complex models).

- **A database of parameters and variables** – for a given flood defence structure, values must be supplied for each parameter and variable required by the relevant LSEs. A value may be fixed or specified as a statistical distribution with associated parameters.
- **A Monte Carlo simulation** – a large sample of input variables (strength and load) are generated and the annual probability of failure, conditional failure probability (where the hydraulic loading conditions are specified as fixed variables and then the strength variables systematically varied) and other related statistics calculated (van Gelder *et al.* 2008). The number of simulations required to achieve a converged estimate of the probability of failure, and thus the calculation time, depend on the chance of asset failure. Most structures in coastal and river engineering, for example, exhibit a relatively high probability of failure (i.e. a relatively low reliability – typically an annual probability of failure > 0.005) compared to structures in other industries where reliability analysis is routinely applied (e.g. for the structural components of a nuclear power plant or mechanical components of an aeroplane the typical reliability will be much higher, < 0.0001). This presents Monte Carlo simulation as a viable and flexible numerical integration tool in the context of the majority of flood defence assets. In the case of complex failure surfaces, where the response of the structure exhibits discontinuities, run times can increase to ensure such discontinuities are captured. In such cases, innovative sampling techniques (e.g. importance sampling) are techniques that could be usefully employed within RELIABLE to minimize run times (van Gelder *et al.* 2008).

In generating the asset-specific fragility information it is important to understand how these methods relate to more traditional assessment methods (based on partial factors of safety). An interesting comparison was completed as part of the Thames Estuary studies (Simm *et al.* 2008), which suggested that a 10% chance of failure can be expected at the design load (reflecting the various safety factors inherent within traditional design) and a 50% chance of geotechnical failure when the factor of safety is equal to 1. These figures are useful **rules of thumb** that can be used to give confidence in the results of the more complex full reliability analysis.

System Analysis and Attributing Risk to Individual Assets

Risk-based management requires a comprehensive consideration of the sources, pathways and receptor impacts. In the context of asset management, this implies that the asset manager must look beyond the performance of individual assets to understand the behaviour of the asset system, and the variation in loading and consequences. System risk analysis, based on the **Source-Pathway-Receptor** concept (DETR 2000; Sayers *et al.* 2002) and methods that sample multiple asset failure combinations (system states) and loading conditions together with the associated impact, are now well established (Hall *et al.* 2003; Sayers and Meadowcroft 2005; Gouldby *et al.* 2008, 2009). Within England and Wales the so-called 'RASP High Level Method' (HR Wallingford 2009) is used in the National Flood Risk Analysis (NaFRA; Steel *et al.* 2009) and is currently being implemented within the Modelling Decision Support Framework (MDSF2; Surendran *et al.* 2008; Environment Agency 2009) and the Performance-based Asset Management System (PAMS; Simm *et al.* 2006; Environment Agency 2010) being developed by the Environment Agency. More detailed system analysis techniques that relax some of the assumptions within the RASP High Level Method are also being developed (e.g. within the Flood Risk Assessment under Climate change, FRACAS; Gouldby *et al.* 2009).

Flood risk system models are typically used to provide a means of quantifying the spatial distribution of risk within the floodplain. For asset management purposes, however, it is desirable to identify those defence sections that make a significant contribution to the risk. This aids the asset manager in targeting and prioritizing resources to maximize risk reduction. Two distinct approaches have been developed to enable risk to be attributed to specific defence assets:

- a rigorous systems modelling approach;
- a simplified field-based approach.

These two approaches are described below.

A rigorous systems modelling approach

The systems modelling approach described by Gouldby *et al.* (2008) enables the contribution that an individual asset makes to the risk to be attributed to that asset. The method of **risk attribution** involves maintaining the relationship between the quantity of water discharged through each individual asset and the quantified impact of the resulting flood. This ability to trace the flow of water across the floodplain is provided by the Rapid Flood Spreading Method (RFSM; Lhomme *et al.* 2008) used within the systems analysis, and enables the relationship between inflow and impact to be explicitly identified for every system state considered (Fig. 15.8).

The attribution of risk to individual assets is achieved by first developing a relationship between the defence assets (d) (as shown in Figure 15.8) and the 'adjacent Impact Zones' (**Impact Zones** are topographic watersheds resolved within the RFSM), and then between 'adjacent' and 'non-adjacent' Impact Zones (i.e. those topographic watersheds remote from the river or coastal boundary).

Through the RFSM it is possible to associate the volume of water discharged into each 'adjacent Impact Zone', under each sampled system state, to the flood depth (hence consequential impact) in other non-adjacent Impact Zones by monitoring the flow of flood water as it propagates across the floodplain area. The quantified impacts associated with each (non-adjacent) Impact Zone are then apportioned to each of the adjacent Impact Zones accordingly (i.e. the total consequential impact for the whole Flood Area is expressed only in terms of the 'adjacent' Impact Zones). The defence contribution, for example for defence number d_1, to the damage c_{d_1} is, for each flooding scenario, simply:

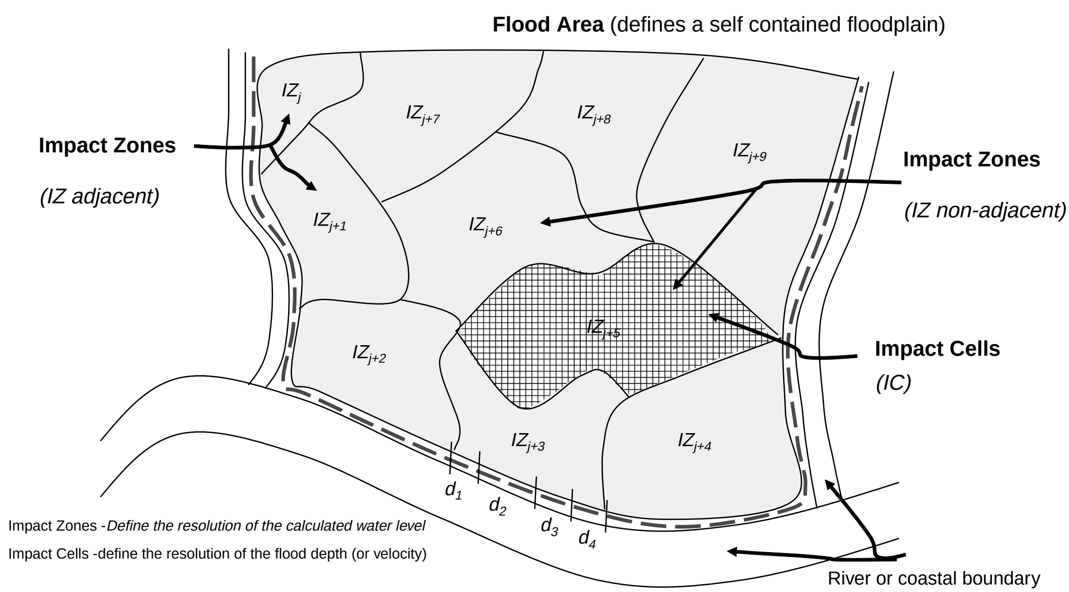


Fig. 15.8 Conceptual diagram of the backdrop of the RASP system model for one Flood Area.

$$c_{d_1} = \frac{v_1}{v_{IZ}} c_{IZ_i} \tag{15.3}$$

where: C_{IZ1} = total economic damage associated with the i^{th} adjacent Impact Zone during the flood event (this equates to the damage within the Impact Zone itself and to those Impact Zones that receive floodwater from the i^{th} adjacent Impact Zone); v_1 = volume discharged into the i^{th} adjacent Impact Zone from defence d_1 during the flood event; and v_{IZj} = total flood volume discharged into the i^{th} adjacent Impact Zone during the flood event.

The defence contribution to the residual risk conditional on the load (l) (based on all system states, m, sampled for load, l) can then be established simply for defence d_1 as follows:

$$\bar{c}_{d_1} l \approx \frac{1}{m_l} \sum_{j=1}^{m_l} c_{d_1} j \tag{15.4}$$

where $\bar{c}_{d_1} l$ = mean consequence associated with the loading condition l for a specified defence (d_1).

The contribution of a given defence asset to the expected annual damage (EAD) is then calculated using the standard formula:

$$EAD_{d_1} \approx \sum_{i=2}^{n} \left[p\left(l_i \geq \frac{l_i + l_{i+1}}{2}\right) - p\left(l_i \geq \frac{l_i + l_{i-1}}{2}\right) \bar{c}_{d_i} l \right] \tag{15.5}$$

where EAD = Expected Annual Damage, and n = number of load conditions considered.

As the volume of water discharged into the floodplain is a function of the defence system state, the state of the defence assets (failed/not failed) is also monitored. The contribution to risk arising from a single defence asset can be further disaggregated into the contribution due to an ultimate limit state failure (i.e. breach in the case of a linear defence asset, or pump or gate failure) or serviceability failure (i.e. overtopping in the case of a linear defence asset, or capacity exceedence in terms of a pump) (Fig. 15.9).

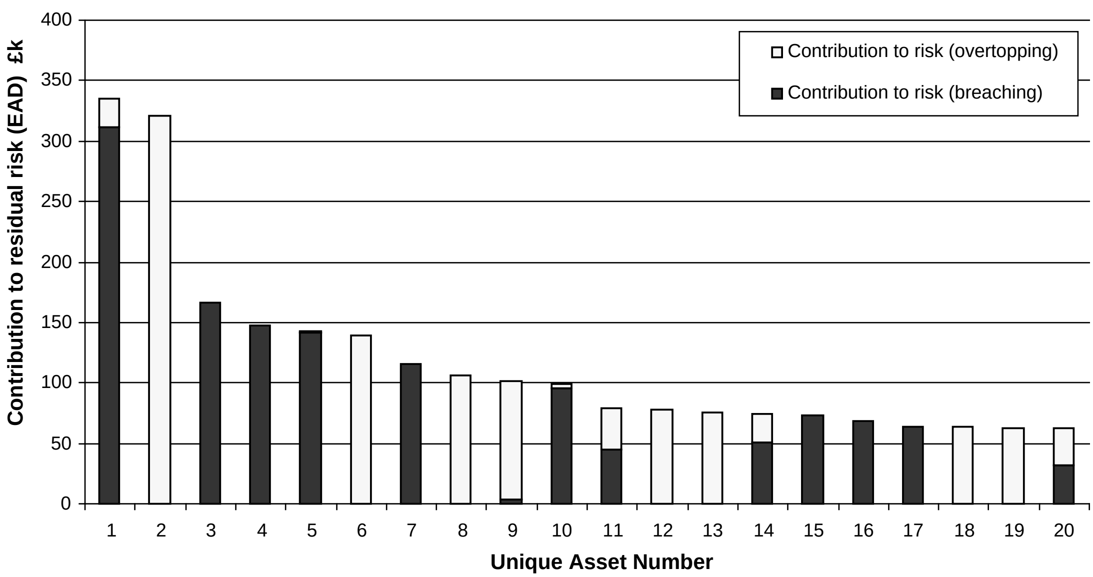


Fig. 15.9 Risk attributed to individual assets can be further disaggregated into the contribution from ultimate (e.g. breach) and serviceability (e.g. overtopping) limit states (taken from the Thames Estuary). Expected annual damage (EAD)

A simplified field-based approach to risk attribution

As well as the rigorous approach to the attribution of risk there is often a requirement to provide a more simplified evaluation based on site inspections (without recourse to complex computational modelling). A simplified tool, 'RAFT – Risk Assessment Field-based Tool' (Environment Agency 2009), can be used to provide a first estimate of:

- the annual probability of failure (breach) – taking account of geometry, structural condition and loading;
- the consequential impacts should a given asset fail – taking into account a range of receptors;
- the risk (taking account of probability and consequence) attributed to an asset in its current condition and assuming improvement to its target condition.

RAFT provides the practitioner with an ability to assess the contribution of an individual asset to risk with a minimum of data and modelling. RAFT uses the physical characteristics of an asset – i.e. crest level and materials of construction – to identify the most suitable high-level RASP fragility curve. It then uses the fragility curve, alongside a user-specified asset length and extreme loading conditions, to estimate the annual probability of failure as follows:

$$\mathbf{P}_{fi} = 1-(1-P_{fCg(i)})^n \qquad (15.6)$$

where $P_{fCg(i)}$ = the annual probability of a single independent section of a given asset i failing (calculated by integrating the fragility curve over all loading conditions); and n = number of independent defence lengths within asset i that can be considered to be Condition Grade j. The number of independent lengths is simply calculated as the total length of the asset divided by either 300 m (for hard defences such as walls and embankments) or 600 m (for soft defences such as beaches and dunes).

In some instances, however, the condition of a single asset may not be uniform. For example, a given asset may have localized problems over a short length with the remainder of the asset in a better condition. In this case, the annual probability of failure can be calculated based on the number of independent lengths considered to be in Condition Grade i and Condition Grade j.

By assuming independence between the part of the asset in Condition Grade i and that in Condition Grade j, a third estimate of annual probability of failure can be estimated:

$$P_{fc} = 1-(1-P_{fi}).(1-P_{fj}) \qquad (15.7)$$

where P_{fi} and P_{fj} represent the annual probability of failure for the proportion of the asset in Condition Grade i and Condition Grade j respectively. The annual probability of failure assigned to the asset as a whole is simply given as:

$$P_f = \max\lfloor P_{fi}, P_{fj}, P_{fc}\rfloor \qquad (15.8)$$

This process ensures that the strength of the asset is not greater than its weakest link (regardless of length) whilst reconsidering that as the asset length increases, so will the chance of failure (assuming all other aspects remain unchanged). Below a limiting length of 300 m the annual probability of failure will be represented by the weakest link within the asset. The influence of asset length on the annual probability of failure is shown in Figure 15.10.

A potential inundation extent is estimated based on the head of water above the floodplain during the event, enabling the user to enter the number of properties that may be inundated in the event of a failure (either estimated or pre-calculated within a geographical information system) and the associated risk calculated taking account of both the probability of failure (i.e. breach) and the associated consequences.

Developing Adaptive and Optimum Intervention Strategies

Often, asset management consists of implementing a range of physical interventions and data improvements staged in time and space, and the asset manager is faced with many difficult questions:

- What is the existing flood risk? Where is it? What are the drivers?
- Which assets contribute the most to flood risk?

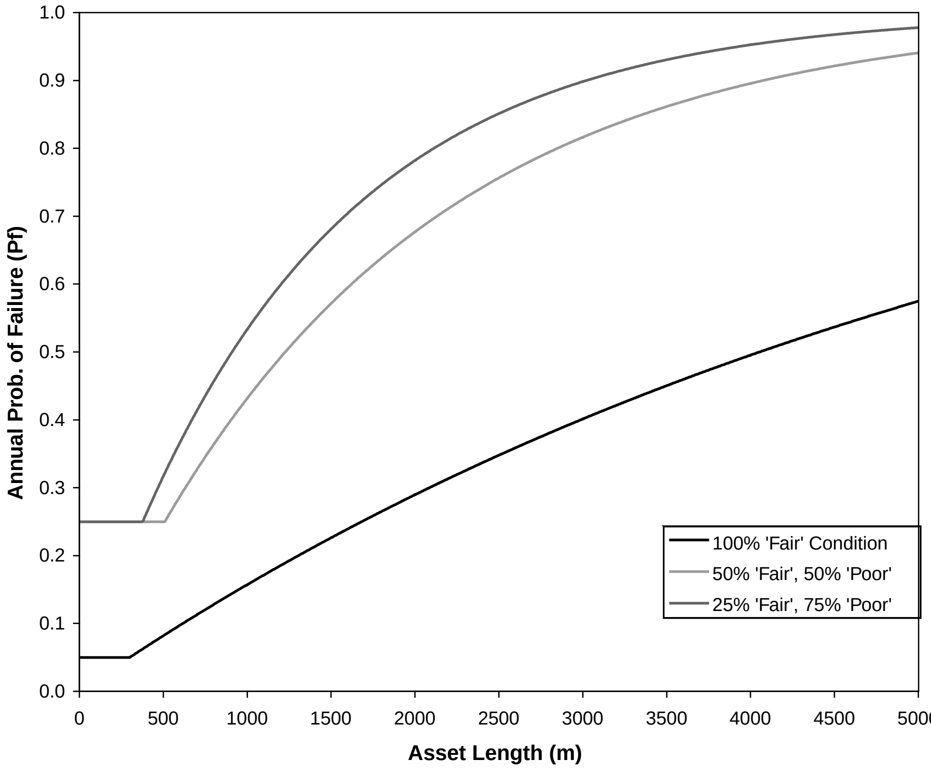


Fig. 15.10 The influence of asset length with the simplified Risk Assessment Field-based Tool (RAFT).

- How would an intervention change the risk?
- How much would a particular intervention cost?
- Is it better to intervene physically or should more data be collected/analysis undertaken? What level of confidence do I have in this decision?
- Which intervention strategy is best, given the future uncertainties?
- Which are the most important uncertainties in terms of their contribution to the doubt as to what to do for the best?

The foregoing sections outline the underlying risk analysis tools that can be used to explore the majority of these questions. More recently a number of projects have started to enact these methods within flexible decision support tools (Surendran *et al*. 2008; McGahey and Sayers 2008; Woodward *et al*. 2010). These decision support tools focus on providing the evidence to support the selection of a preferred investment action/strategy. Various considerations are essential to this process beyond the analysis of flood risk, including:

- **Robustness:** are the proposed actions robust? – i.e. will the strategy perform well in the context of a wide range of possible futures (resources, climate, socioeconomic, etc.)?
- **Flexibility:** are the proposed actions flexible? – i.e. are future choices compromised, or can alternative actions be taken at a future date with limited additional cost?
- **Adaptability:** can the strategy be adapted to account for future change? – i.e. as the reality of the future unfolds can the asset designs be adapted (heightened/widened, etc.) with minimal cost?
- **Uncertainty:** is the strategy robust to the uncertainty within the data, methods and model structures, as well as to the gross uncertainty associated with future change?

To use these criteria in assessing the likely impact of the investment, they must be considered not only in broad terms reflecting socioeconomics and climate change, but also in terms of budgetary and legislative constraints and environmental impacts and opportunities. Understanding the robustness, flexibility and adaptability of an asset management strategy in quantifiable terms, however, remains the more elusive aim of sustainability in this context.

Supporting tools and techniques in aiding robust option choices

In selecting the **best** investment strategy the decision-maker is faced with choosing between many possible options of physical intervention, further data collection and analysis. Underlying this choice is a desire to maintain the flood risk system's ability to perform reasonably well in the context of all plausible futures that may be encountered throughout the appraisal period (i.e. funding changes and future affordability, climatic conditions, changes in anticipated performance).

In this context **performance** is typically measured in terms of efficiency (e.g. risk reduction, opportunity benefit) and effectiveness (e.g. benefit to cost ratio). Determining the preference ordering, assuming prefect information, would be a straightforward ranking process. But a multiple of both aleatory and epistemic uncertainties combine to complicate this process.

Classical decision theory (e.g. French 1988) discusses two widely considered approaches to deal with such uncertainty. One, based upon Laplace's Principle of Indifference or Insufficient Reason, involves assigning an equal probability to uncertain quantities; and is therefore fundamentally probabilistic. The other, Wald's Maximin model, makes the assumption that the worst case of the uncertain quantity will always arise, and seeks to choose the option that maximizes the reward given this assumption – the approach does not therefore involve assigning any likelihood to uncertain quantities.

'More recently Info-Gap approaches, that purport to be non probabilistic in nature, developed by Ben-Haim (2006) have been applied in the context of flood risk management by Hall and Harvey (2009). Sniedovich (2007), is critical of info-gap approaches suggesting the approach is based upon analysis in the neighborhood on a point estimate of the system state (the uncertain phenomena) and the output of the analysis is sensitive to this decision. The method makes the assumption that the future system states become increasingly unlikely as they diverge from the point estimate (Hall (2009)). The method assumes that the most likely future system state is known a priori. Given that the system state is subject to severe uncertainty, an approach that relies on this assumption as its basis appears paradoxical and this is strongly questioned by Sniedovich (2007)'.

A more traditional method that involves Bayesian type probabilistic weighting for future scenarios and incorporating these into analysis of options has been explored through application to the River Thames, including:

- **Intervention scenarios/decision pipelines** (Fig. 15.11) – This includes analysis of a limited range of expert-derived **decision pipelines** that describe a logical progression of management choices that are constrained by the preceding choices. Each decision point is constrained by previous actions and as such is more or less suited to different future states. Such analysis provides a pragmatic means of developing and exploring future asset management options (McGahey and Sayers 2008).
- **Formal optimization of the asset intervention investment strategy** – More automated methods to optimize an asset management strategy have recently started to appear in the context of flood risk management (Philips. 2006; Woodward *et al*. 2010). These methods draw on various fields within civil engineering (including bridge maintenance, truss design and pipe network design) where optimization methods are more widely used. The most promising methods are based around genetic algorithms (GAs), reflecting their ability to optimize performance across the many criteria of interest associated with flood risk management decisions.

Genetic algorithms work by seeking to combine the desirable qualities from solutions already

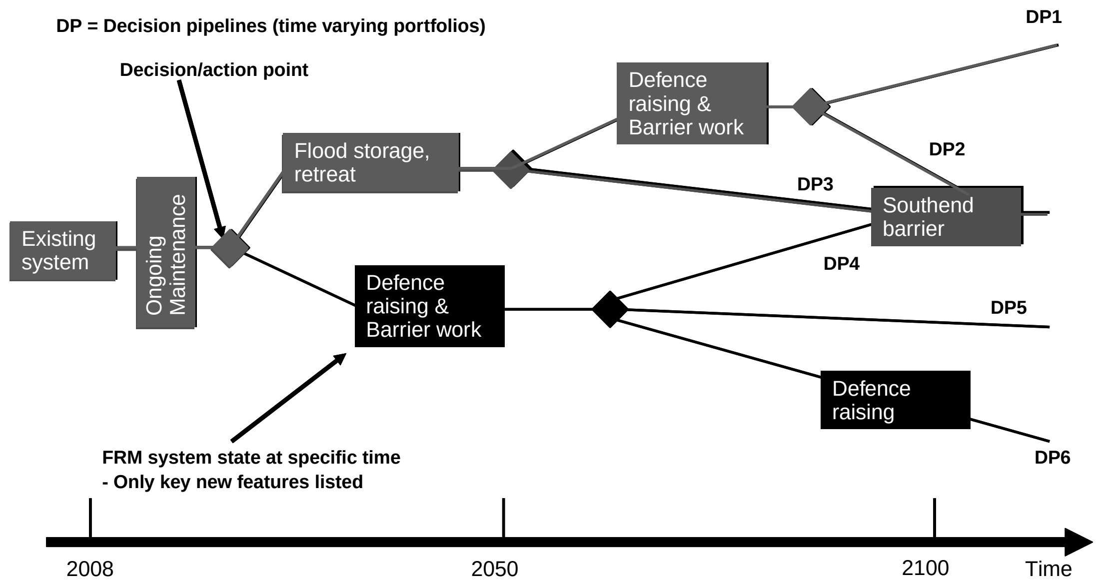


Fig. 15.11 The performance of different strategic alternatives (represented by unique routes through the future decisions) enable adaptive strategies to be developed that reflect future uncertainty. FRM, flood risk management.

found to create solutions that are even more desirable. In this way its search of the solution space is not as regimented as hill-climbing techniques (allowing the search to be targeted in areas thought to be most favourable, whilst not restricting the search based on this bias) nor as random as Monte Carlo sampling. By combining the characteristics of two different 'parent' solutions a new solution is proposed. The best-performing offspring are selected and recombined to develop (hopefully) ever **fitter** solutions. This process is repeated over several 'generations' (iterations) until the maximum utility (across multiple criteria) of the solution is reached.

Tools to optimize basic interventions (crest level raising, condition grade improvements) have now been trialled on simple flood risk management studies (Philips. 2006; Woodward *et al*. 2010). The basic building blocks of these tools are shown in Figure 15.12 and include:

• *A description of autonomous future changes:* The future is of course unknown and there are many uncertain influences outside of the asset manager's control. For example (i) **climate change** (UK Climate Programme 2009, UKCP09) provides a probabilistic description of potential future climates than can be readily utilized within the optimization process); (ii) **asset deterioration** [in the absence of management or reduced management. Expert-based deterioration curves are typically used to describe deterioration from one condition grade to the next (Simm *et al*. 2008). Process-based statistical models are starting to emerge with the ability to model asset time-dependent processes using Markov processes, such as the Poisson or the gamma process (Buijs *et al*. 2005)]; (iii) **central budgetary change**; and (iv) **socioeconomic change and floodplain development**.

• *An ability to incorporate multiple (competing) objectives:* Flood risk management takes place in a world of many competing demands. Optimization allows these to be explicitly described as objective functions, for example (i) maximise economic benefits; (ii) minimize whole-life costs; (iii) minimize loss of life; (iv) maximize environmental enhancements, etc. whilst, for example, reflecting budgetary constraints.

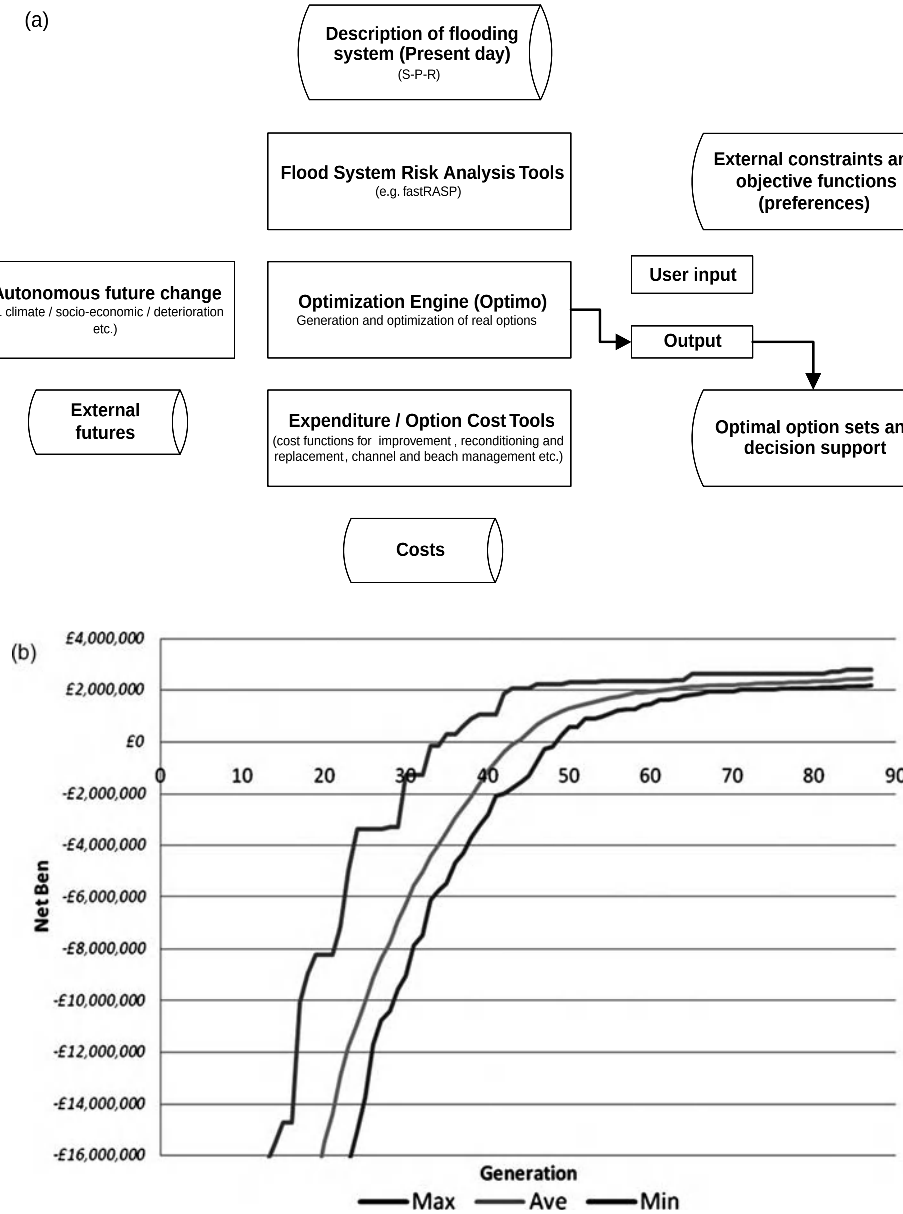


Fig. 15.12 (a) Building blocks of an optimization tool being developed by HR Wallingford/Exeter University. (b) Nett Benefit of the investment strategy proposed by each successive generation (Philips, 2006).

- *An ability to explore real options:* Real options describe 'a choice that becomes available through an investment opportunity or action' (HM Treasury 2009). GAs enable the development of optimal, real options through time that take account of the constraints placed upon future choices by previous actions. This enables options to emerge that are appropriately adaptive and robust to future uncertainty – whilst achieving maximum utility when judged against the range of user-defined objective functions. Practical working methods can also be reflected and included within the GA, such as annual budgetary ceilings, or perhaps practical working constraints such as addressing multiple issues once the work force is mobilized. The progress of the GA in finding the best solutions (i.e. ones with higher net benefits) can be seen by plotting the best solution found for each generation (Fig. 15.12). As shown in Figure 15.12 the same optimum strategy (in this case based on a single objective function of Nett Benefit) emerges after around 60 generations.

Review, Decide and Act

Asset management tools seek to provide evidence in support of decisions, but not, of course, to make decisions. Expert judgement and engineering skill will continue to feature strongly throughout the asset management process – from the input data through to confirming the preferred course of action. Incorporating the expert judgement in an unbiased and transparent manner is problematic. Considerable progress has been made in recent years to integrate expert judgement and quantified analysis tools (Simm *et al.* 2008; Hall and Solomatine 2008). In particular expert judgement can be used to validate model inputs and provide credibility to (and validation of) the outputs from the analysis. The decision-maker also needs to be confident that the decision made is robust to the uncertainty in the data, the predicted impact of the action (e.g. reduced risk) and the associated cost (e.g. whole-life costs and benefits). Quantified uncertainty propagation methods (Gouldby *et al.* 2010) together with multi-criteria decision-making provide efficient methods to support the decision-maker in identifying robust choices. Although these are interesting and important areas they are not discussed further in this chapter.

Future Uptake: Barriers and Facilitators

The introduction of formal and structured risk-based approaches to asset management challenges many traditional ideas and can be difficult to achieve in practice. Many of the barriers to the uptake of such methods reflect capacity to adopt new approaches, misconceptions around the complexities of risk-based methods and the challenge of converting good science into practical usable tools. The science and practice of asset management need to go hand in hand – with one evolving from the other. The capacity for change is limited (in both skills and supporting infrastructure) and the scientific demand for change must be commensurate with the practical benefits afforded by that change. Simple illustrations and pilot studies that explain the complex scientific processes in practical terms are a vital aid in building understanding in the user community and avoiding mistrust and misconception.

Conclusions

The implementation of risk-based asset management reflecting whole-life performance will demand close collaboration between the science community and engineering practice. To be successful there are a myriad of activities that will need to be integrated and coordinated within and outside of those organizations with a direct interest in managing flood defence assets. As this chapter highlights, system analysis, reliability, risk attribution and optimization techniques do, however, provide a number of important insights and aids to the decision-maker.

The RELIABLE analysis tool provides a flexible and practical means to analyse the reliability of most structures. The results from RELIABLE have been shown to be credible and easy to apply, enabling high-level generic fragility curves to be replaced (within the system analysis models) where the need is greatest.

An understanding of an asset's chance of failure (now and in the future) is an important contribution to understanding the risk and how best to manage it; but it is not the only consideration. Assets must be understood in the context of the asset system within which they reside. It is important: (i) to consider a full range of inundation scenarios (with and without one or more asset failures) across a wide range of storm events (from the frequent to rare); (ii) to evaluate the potential associated impacts (economic as well as other damages and importantly opportunities); and (iii) to integrate the results accordingly. Credible system analysis methods are now available and embedded within various tools. These tools are capable of attributing risk to individual assets, which in turn provides a powerful support to the identification of critical defence assets.

The trial application of the more formal methods to optimize asset management strategies indicates that GAs are able to efficiently search the option space, and react appropriately to any constraints on the desired solution (budgetary, environmental, safety, etc.). Computational and complexity limitations continue to restrict the implementation of GAs more widely for practical decision support. At present expert-led development of real options, for example through the use of decision pipelines, will continue to offer a very credible stop-gap.

Information technology is at the heart of an efficient approach to asset management (supporting the principles of good asset management). The USACE in the US, the Rijkswaterstaat in the Netherlands and the Environment Agency in the UK have all undertaken similar initiatives to improve the underlying data and access to it.

All asset managers want to be more efficient and more effective in their decision-making. The advances outlined in this chapter afford significant future opportunities for the asset manager to achieve this through a better understanding of the individual assets and the asset systems they comprise.

Acknowledgements

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16 Handling Uncertainty in Coastal Modelling

DOMINIC E. REEVE, JOSÉ HORRILLO-CARABALLO AND ADRIÁN PEDROZO-ACUÑA

Introduction

'Why should we bother to model coastal processes?' is a question often asked by practitioners. It is a natural question, which has a straightforward answer: for the same reason that we try to forecast future weather, stock market movements, and the outcome of sporting events. That is, to try to gain warning of dangerous events, to make a fortune or to gain an edge over the competition. In coastal engineering, modelling is used as part of the process of designing flood defence or coastal protection works to withstand the conditions that they are likely to experience over the period for which they are intended to operate. In this chapter a range of modelling techniques used in coastal engineering is illustrated via a set of case studies. Particular attention is given to ambiguities accompanying the modelling, and methods for quantifying this uncertainty.

Modelling is the process through which predictions of the future are made. The modelling process may not attempt to provide a detailed forecast of actual future events, but might be aimed at predicting the outcome under particular 'worst case' scenarios. Modelling can take many forms, typically involving quantitative methods such as theoretical analysis or computational simulation.

For coastal applications, the main quantities required for design are a description of the wave conditions, the water level variations and the movement of sediment. For waves, rather than describing the sequence of individual waves it is common practice to provide a statistical description of the conditions that define representative wave heights, wave periods and wave directions. For water levels, the situation is slightly different. In many locations around the world the predominant component of water level variations at the coast is due to tides. Tides result from the gravitational attraction between the Sun, Moon and Earth, are well understood and are predictable to a good level of accuracy. An additional, and less predictable, component to the water level is 'surge'. This term describes changes in the water level arising from meteorological effects. Sediment transport is not fully understood and is currently the subject of intense research activity. Some of the factors that make this such a difficult area include: the range of sediment particle shapes and sizes; the very different responses of cohesive sediments, sands and gravels to the same wave conditions; the three-way interaction between the shape of the seabed, the incoming waves and the prevailing water levels; the vast difference in timescales between the period of an individual wave and the period over which significant changes in beach morphology occur; and, similarly, the large range of spatial scales over which sediment transport and morphological changes occur.

Forecasting methods have evolved as our understanding of the physical processes has

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improved, and as computer power has developed. Thus, the use of charts, tables and hand calculations has gradually given way to computer models.

The search for better understanding to improve our predictions has led to greater monitoring of waves, tides and beaches. An important agent for this in the UK was the introduction of Shoreline Management Plans in the early 1990s, which set a requirement on data gathering for coastal management and development. Thus, there are now a growing number of coastal locations for which there are records covering over 10 years. While this is not yet adequate for detailed research, the dataset does provide valuable information for planning and design.

In all prediction it is important to acknowledge uncertainty. Sources of uncertainty include:

• Incompleteness – if not all relevant processes are taken into account then the predictions are unlikely to be realistic.
• Empiricism – design equations are generally empirical, based on experiments performed at laboratory scale. The measurements have scatter leading to errors in fitting an equation to the data.
• Extrapolation – when estimating design conditions some extrapolation is usually required and there is uncertainty associated with this.
• Measurement error – the observations used for design will have uncertainties due to the limitations of the measurement equipment.
• Non-stationarity – if there is a long-term underlying trend (such as a gradual rise in the mean level of the sea), or if the variance of a quantity changes over time (such as storm intensity or duration), then due account must be made for this.

While some uncertainty has to be accepted its effects can be mitigated to some extent by considering 'worst case scenarios', including factors of safety based on engineering judgement, and, where appropriate, by adopting a probabilistic approach to design that allows uncertainties to be quantified (e.g. Thoft-Christensen and Baker 1982; Melchers 1999; Reeve 2010). In the following sections the basic categories of modelling techniques are described, with some case studies to illustrate the methods.

Modelling and Prediction Techniques

One of the first decisions to be taken is what model to employ. This will depend on the type and amount of data available, as well as commercial considerations. The data will usually be in the form of:

• time-series (values at a fixed location at regular intervals in time);
• seasonal or annual statistics;
• average or 'typical' conditions;
• specified conditions corresponding to a given return period;
• qualitative information on past construction of sea defences, beach nourishment and dredging operations.

Models for describing coastal processes may be categorized into four types:

1 Statistical (based on analysis and extrapolation, requiring long records of observations).
2 Empirical (describing equilibrium conditions, requiring minimal information).
3 Dynamical (solving the equations of motion, requiring a moderate amount of data).
4 Hybrid (mixtures of the above types, requiring a modest amount of data).

Examples of methods from each of these categories are given below.

Statistical analysis and extrapolation

Statistical techniques are applied to time series of waves and beach level measurements to identify patterns of behaviour. This approach can be successful if there is strong periodic behaviour in the data. Fourier analysis is often the method of choice. However, if the spatial or temporal sampling is at irregular intervals then interpolation to regular intervals will be necessary. The Empirical Orthogonal Function (EOF) technique allows patterns to be identified from irregularly sampled data and has been used extensively (e.g. Winant *et al*. 1975; Reeve *et al*. 2001; Miller and Dean 2007). However, extrapolation into the future is not so straightforward. Both Fourier and EOF methods are based on an assumption that the data record is statistically stationary. If this is not the case then alternative methods or some pre-filtering of the data are necessary. To determine design conditions, techniques are available to fit probability distributions to measurements. Extreme values corresponding to the chosen return period are obtained by extrapolation (e.g. Reeve *et al.* 2004).

All statistical methods are dependent on good and extensive measurements. Also, extrapolation into the future is usually made on the basis that past behaviour is a good indicator of future evolution, and will therefore be unable to capture behaviour absent from the data.

Empirical models

Empirical models are usually based on equilibrium type arguments, and describe the shape of the beach under particular (unchanging) wave conditions. In reality, wave conditions change continually, but the predominant conditions can provide a useful guide to the 'typical' beach shape. Bruun (1954) found that many ocean-facing coastlines exhibit a concave curve that can be described by the equation:

$$h = Ax^{2/3} \tag{16.1}$$

where h is the profile depth at a distance x from the shoreline, and A is a constant, which has been related to grain size, D, by Dean (1991), i.e. $A = 0.21D^{0.48}$ with D in mm. These equations predict that equilibrium beach slopes increase in steepness with increasing grain size, in accordance with observations. On beaches that are not well sorted, i.e. with a range of grain sizes, there can be uncertainty over the appropriate value of D to use. Further, this model can describe neither the temporal evolution of a beach, nor the formation of bars and troughs. Figure 16.1 shows an example of fitting Equation 16.1 to a measured beach profile in Colombia. The equilibrium curve fits the general trend of the measurements reasonably well, but there are some notable (and large) discrepancies both nearshore and further offshore.

Bays form where an erodible coastline exists between hard, stable headlands. The shape of the bay will depend on the wave climate and supply of sediments. Silvester (1974) performed laboratory experiments to investigate the equilibrium shape of bays for different wave conditions. He found that in the absence of sediment supply and for a fixed wave direction, a stable bay would take the approximate form of a cardioid; the beach adapting its shape so that the incoming wave crests, which are curved due to diffraction, are everywhere parallel to the shore (Fig. 16.2). More recent work on this method may be found in Hsu *et al.* (1989), Silvester and Hsu (1997), and Gonzalez and

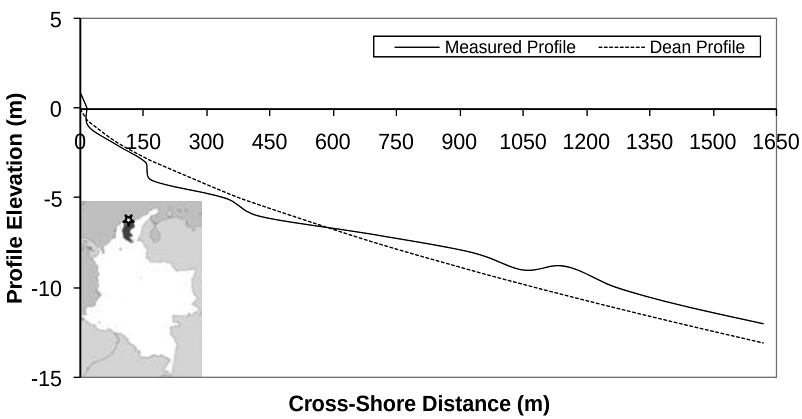

Fig. 16.1 Comparison of Dean profile and measured profile of a beach in Santa Marta, Colombia (South America).

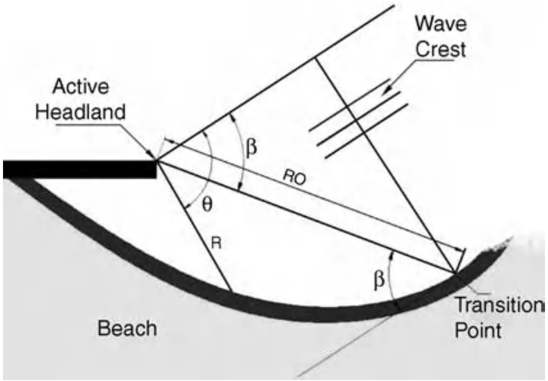

Fig. 16.2 An example of a headland bay and the corresponding equilibrium shape. β is the angle between the radius to the transition point and the incoming wave crests; υ is the reference angle, which varies from β upwards to the maximum required to describe the shape of the bay; R is the length of the radius from the headland to the beach line in the bay; R_0 is the baseline radius – the length between the headland and the transition point; the transition point is the location on the beach where the tangent to the beach is parallel to the incoming wave crests.

Medina (2001), who discuss some of the uncertainties surrounding this technique.

Due to their generic nature these models have wide applicability; they provide a simple way of predicting the stable bay shape and the typical profile. Fluctuations from these forms arise from the fact that beaches are rarely in strict equilibrium.

Deterministic process models

These are computational models that solve the equations of motion expressing conservation of mass and momentum for water, and mass conservation for sediment. They include detailed descriptions of the sediment transport process, including suspension, transport and settling. The models are iterative, requiring sequential solution of the hydrodynamics, sediment transport (q) and bathymetric updating. Time steps for hydrodynamics (Δt) are usually much shorter than for the bathymetric updating (Δt_{morph}). Hence, bathymetry is often considered fixed for the hydrodynamic step until a 'significant' change occurs. At this point the bathymetry is updated and then the hydrodynamics runs for the updated bathymetry. There are many uncertainties in the sediment transport formulae, as well as cumulative errors in the iterative scheme. These models can be prone to instability due to feedback between hydrodynamic and bathymetric changes. This is usually solved by controlling the bed steepness through an 'avalanching' step to prevent unrealistically steep slopes developing in the predictions. An example of the structure of a dynamic process model is shown in Figure 16.3.

Hybrid models

This type of model often employs a simplification of the physical processes into a few evolutionary equations (e.g. Stive and De Vriend 1995;

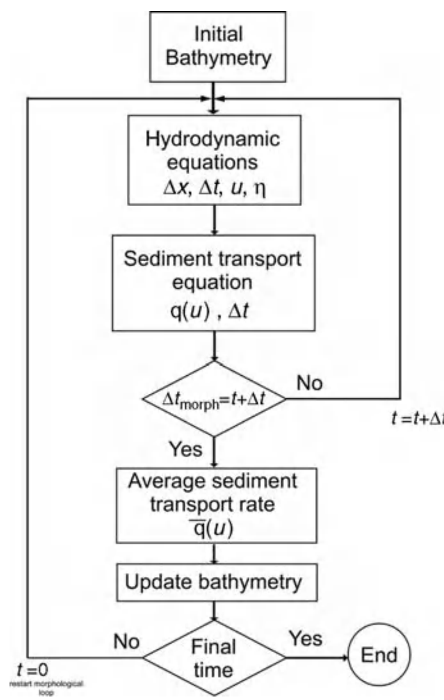

Fig. 16.3 Example structure of a detailed process model of coastal morphology.

Larson *et al.* 1997; Van Goor *et al.* 2001; Karunarathna and Reeve 2008). Predictions can be made on the basis of parameterizing all but a few processes as a source function. These models have had reasonable success in predicting changes in morphology but there is no established method for defining the parameterization.

Case study: Tidal Flow Prediction

A tidal flow model, developed by Osment (1992) and based on the approach of Falconer (1976), is used to calculate the tidal flow in the southern North Sea. This model simulates unsteady two-dimensional flows in one-layer (vertically homogeneous) fluids.

The numerical model uses the finite difference Alternating Direction Implicit (ADI) technique to integrate the equations for mass and momentum conservation. This method is a well-proven technique that has the ability to handle moving boundaries. There are various implementations of the finite difference method that give different levels of accuracy and stability. Here, the hydrodynamic equations are solved with a central approximation for second-order accuracy and an implicit scheme. The solution is performed over the southern North Sea area shown in Figure 16.4.

Model set-up

The open boundary in the south is located between Rye in England and Cap Gris Nez in France. The open boundary in the north is located between Flamborough Head in England and Norderney in Germany. The bed elevations vary between 2 m above Chart Datum (CD) and 95 m below CD in Silver Pit in front of the Humber Estuary. The sea bottom shows different forms and features including sand waves up to 16 m high. There are also numerous tidal sandbanks in the area (e.g. the Norfolk, Great Yarmouth, Flemish and Zeeland Banks).

The selection of the boundary conditions satisfied the requirement that there are some tidal stations that can provide data for comparison with the results obtained by the model. The digital

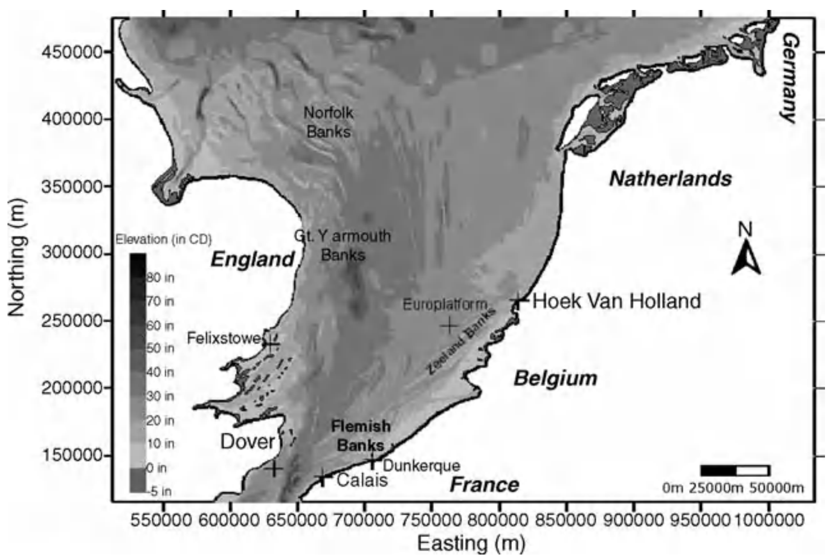

Fig. 16.4 Model domain and southern North Sea bathymetry in the UK National Grid reference system digitized from the Admiralty Chart No. 2182A showing the position of the calibration and validation points.

terrain model (DTM) or model topography was generated on a staggered finite difference grid with a spatial resolution of 1045 m. Boundary conditions were specified by imposing the amplitudes and phases of preselected tidal constituents at the nodes of the open boundaries. Along the open boundaries the harmonic constants are available for five constituents: O_1, K_1, N_2, M_2 and S_2. The amplitudes and phases for these constituents were obtained from Howarth (1990).

A time step of 120 seconds was used and the model was run for a period of 60 days from 1 January 2001 to cover four spring-neap cycles, in order to acclimatize the model to the boundary conditions and to have a good representation of the flow dynamics in the area. Further details of the modelling can be found in Horrillo-Caraballo (2005).

Model calibration and validation

The main sources of uncertainty arising in the model are: numerical errors due to the choice of grid size and time step; the representation of frictional effects; and errors in the bathymetry and boundary conditions. To quantify these errors a process of model calibration and validation is performed (Horrillo-Caraballo 2005). To calibrate the southern North Sea model, a range of cases were studied. First, three different grid resolutions (2090 m, 1045 m and 522 m) were used to choose the best grid spacing. The 1045-m grid was the best option for these studies giving good results in reasonable computing time. A second set of tests was performed in which the bed stress was changed by altering the Nikuradse coefficients (k_s). The influence of the bottom friction on the model's ability to reproduce observed data was examined first. The computed sequences of surface elevations at each grid node over 60 days were subjected to harmonic analysis. For each constituent, a grid map was made in order to compare it with cotidal maps derived from extensive observations (Howarth 1990).

According to the calibration for the southern North Sea, positions of amphidromic points are well captured and constituent amplitudes agree, in general terms, to within 10% (< 7 cm) or better, and constituent phases agree on average to within 5% (< 16°) or better throughout the model domain. Figure 16.5 illustrates the calibration process, and Table 16.1 summarizes the best level of agreement obtained with $k_s = 0.05$.

Values computed by the model were compared with corresponding values predicted from tide tables from the British Oceanographic Data Centre (BODC), and the French Tide Table, (FTT). Figure 16.6 shows the computed and predicted tide levels at Felixstowe and Dunkerque.

The model demonstrated the same level of percentage error against the validation test as in the calibration. The average error in the phase of the harmonic constituents is less than 16° (< 5%) and the average error in the amplitude of the harmonic constituents is less than 7 cm (< 10%). The calibration and validation procedure provides an objective means of quantifying the uncertainties in complex model outputs.

This case illustrates how the magnitude of the uncertainty in a variable (the tidal amplitude) can be investigated through sensitivity testing by changing the value of key parameters. It also shows how a numerical model could be used in a more formal Level 2 type probability analysis. For example, we might be interested in the spring high tide level at a particular port. The three results for different values of bed friction coefficient could be used to estimate the derivative of tide level with respect to the coefficient, which is needed for the Mean Value Approximation method (see, e.g., Melchers 1999; Reeve 2010) for estimating the probability of failure.

Case Study: Statistical Modelling of Beaches

The presence of a healthy beach is often an integral component of modern coastal flood defence design. In addition, it is important from a designer's and shoreline manager's perspective to be able to predict beach behaviour with some level of confidence. Numerical models are available for this but can be difficult to operate. An alternative to process-based modelling is

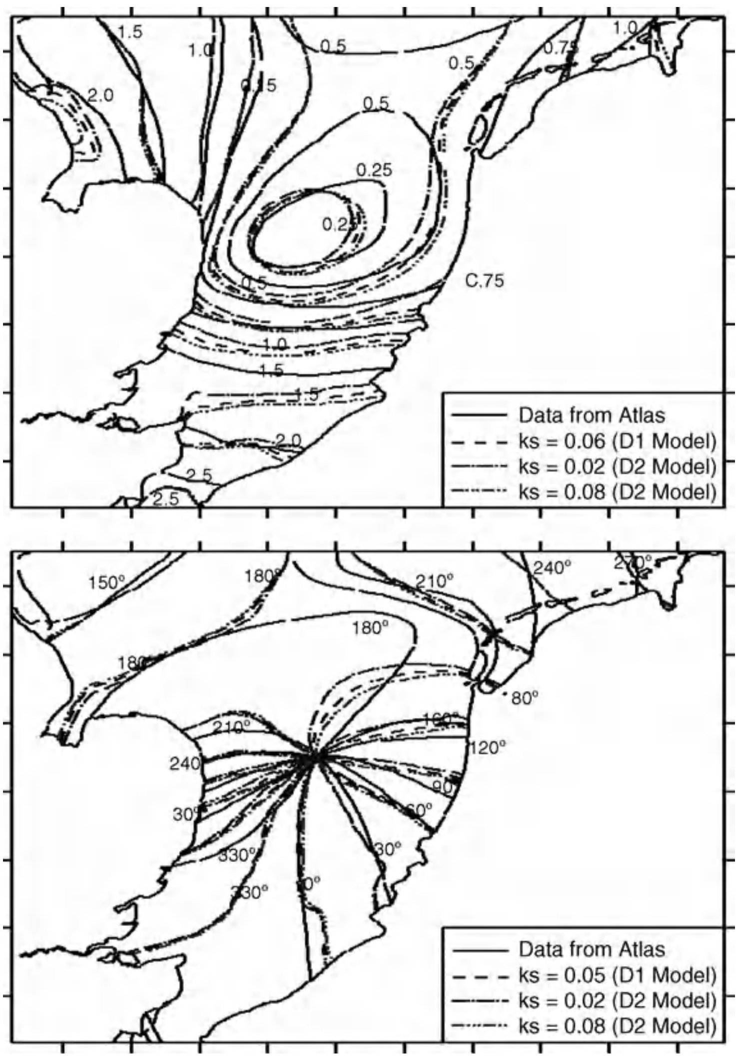

Fig. 16.5 Comparison between computed (a) amplitude and (b) phase and data from co-tidal charts (Howarth 1990) of the harmonic constituent M_2 for different bed friction coefficients.

data-driven modelling. In essence, this relies on a statistical analysis of historical data and extrapolation.

One of these techniques is the Canonical Correlation Analysis (CCA). CCA measures the relationship between the observed values of two sets of variables. This method has been used in different fields, such as meteorology and climatology (Barnett and Preisendorfer 1987; Graham *et al.* 1987; Shen *et al.* 2001). It has also been used in the discipline of coastal engineering to detect patterns in the wave and profile data, to predict the beach profile response due to waves (Larson *et al.* 2000) and to analyse the evolution patterns of multiple longshore bars and their interactions (Różyński 2003).

Here, wave measurements and beach profiles were used to extend the analysis of the beach at Duck, North Carolina, by Larson *et al.* (2000) over a longer period. The data covered beach profile surveys taken along fixed profiles normal to the beach line, and spectral wave properties

Table 16.1 Amplitude (A) and phase (Ph) relative to Greenwich (rounded to the nearest degree) from the Tidal Atlas (Howarth 1990), model results and their difference

	O_1		K_1		N_2		M_2		S_2	
	A (m)	Ph (deg)	A (m)	Ph (deg)	A (m)	Ph (deg)	A (m)	Ph (deg)	A (m)	Ph (deg)
FELIXSTOWE										
Data from Atlas	0.13	174	0.10	354	0.19	300	1.20	320	0.32	20
Model	0.12	175	0.10	2	0.17	297	1.10	321	0.30	22
Difference	0.01	−1	0	−8	0.02	3	0.10	−1	0.02	−2
DOVER										
Data from Atlas	0.06	165	0.06	45	0.38	310	2.29	330	0.70	28
Model	0.04	140	0.04	63	0.44	305	2.33	310	0.80	10
Difference	0.02	25	0.02	−18	−0.06	5	−0.04	20	−0.10	18
CALAIS										
Data from Atlas	0.05	135	0.02	70	0.42	330	2.40	340	0.75	39
Model	0.04	140	0.04	63	0.44	305	2.33	310	0.80	10
Difference	0.01	−5	−0.02	7	−0.02	25	0.07	30	−0.05	29
DUNKERQUE										
Data from Atlas	0.08	154	0.04	30	0.35	334	2.10	352	0.62	50
Model	0.07	152	0.04	8	0.37	325	2.24	335	0.71	37
Difference	0.01	2	0	22	−0.02	9	−0.14	17	−0.09	13
H.V. HOLLAND										
Data from Atlas	0.10	180	0.08	15	0.10	45	0.85	60	0.20	120
Model	0.11	172	0.08	356	0.10	30	0.83	45	0.20	113
Difference	−0.01	8	0	19	0	15	0.02	15	0	7
EUROPLATFORM										
Data from Atlas	0.11	178	0.09	4	0.11	3	0.77	20	0.14	105
Model	0.11	172	0.08	358	0.10	1	0.84	19	0.22	83
Difference	0	6	0.01	6	0.01	2	−0.07	1	−0.08	22

(significant height and peak period) recorded at least every 3 hours during the same period. The wave measurements were obtained from *in situ* wave recorders.

Dataset

The Field Research Facility (FRF) is located on the Atlantic Ocean in Duck, North Carolina, USA. It is managed by the US Army Corps of Engineers to support the Corps' coastal engineering research. Since the creation of the FRF, a comprehensive long-term monitoring programme of the coastal zone, including waves, tides, currents, local meteorology and beach response, has been maintained.

Measurements from this site have been used to refine theories of the nearshore morphological changes and to develop and test numerical models of nearshore response to natural conditions (Plant *et al*. 1999; Howd and Birkemeier 1987; Reeve *et al*. 2007).

Beach profile dataset

The data used for this study were from the profile Line 62, obtained from the FRF website (http://www.frf.usace.army.mil/frf.shtml) and cover the period from July 1981 to December 2003 (Fig. 16.7a). Each profile was interpolated every 2 m, using a cubic spline interpolant to keep the concavity of the raw data (Li *et al*. 2005). The interpolation was carried out from the dune level at around 70 m from the main baseline until 910 m offshore (around 8 m water depth). For this case 225 profiles were used.

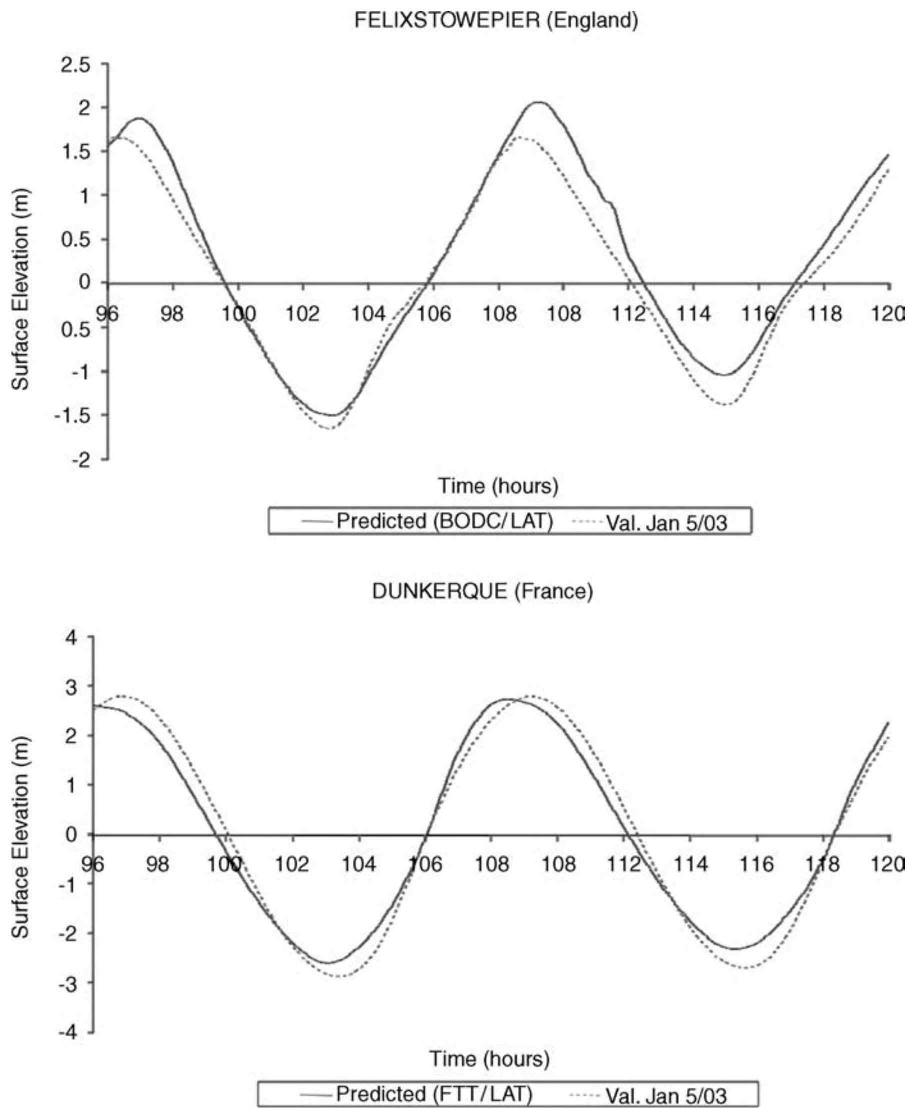

Fig. 16.6 Comparison of predicted and computed time-series elevation for 24 hours starting 5 January 2003, at 0000 hours referred to the lowest astronomical tide (LAT). BODC, British Oceanographic Data Centre; FTT, French Tide Table.

Wave dataset

The wave data consisted of significant wave height (H_s) and peak spectral wave period (T_p) obtained from the instruments deployed in the FRF. The data were collected from an array of 15 pressure gauges, referred to as gauge 3111, located approximately 1 km offshore and at 8 m water depth. Wave height and period were typically recorded every 6 hours but more frequently during some parts of the observation period, for which hourly values were recorded. The dataset used in this study covers the period from 1981 to 2005.

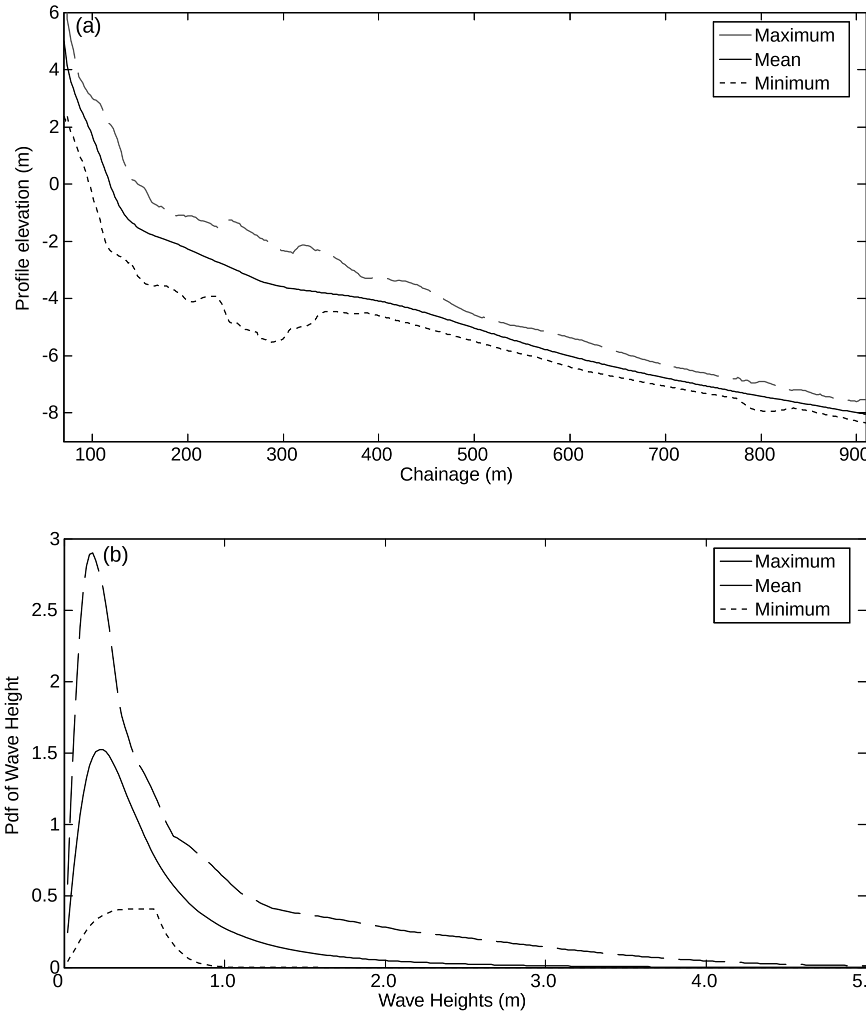

Fig. 16.7 (a) Mean, upper and lower envelope of measured beach profiles; (b) mean, upper and lower envelope of Rayleigh wave height probability density functions (pdf) used as input in the canonical correlation analysis (CCA).

Rayleigh distribution

To represent the wave climate, probability density functions (pdfs) were derived from the wave heights. The pdfs of waves between two successive beach surveys were assigned to the later survey. The physical justification of this is that the changes in the beach profile represent the cumulative effect of all the waves impinging on the beach since the previous measurements.

The Rayleigh distribution has proven to be a reasonable measure of the wave height distribution for waves in deep water and is defined as:

$$p_N = \frac{1}{N}\sum_{i=1}^{N}\frac{2H}{H_{rms0,i}^2}e^{-\left(\frac{H}{H_{rms0,i}}\right)^2} \tag{16.2}$$

where p_N is a combined probability density function of wave heights that is derived by superimposing the individual pdfs available for the measurement period between surveys; N is the number of individual wave measurements between surveys; i is an index; H is the wave height; and H_{rms0} is the root-mean-squared wave height in deep water (with $H_{rms} \approx 0.707 H_s$, and the '0' subscript referring to quantities measured in deep water or 'offshore').

The superposition carried out in Equation 16.2 implies that all the individual pdfs derived from the wave measurements (H_{rms0}) have the same weight. Figure 16.7b shows the corresponding composite pdfs for the nearshore wave height valid for the time period between surveys and obtained by summing over a large number of Rayleigh pdfs according to Equation 16.2.

Predictions of profile changes based on Rayleigh distribution

The data were represented by truncated forms of their EOF expansions in order to reduce the noise in the records, before performing the CCA analysis. In general, three to five EOF modes were sufficient to represent most of the variation in the datasets. In this study, five EOF modes were used but only three are plotted. Table 16.2 summarizes the results for the first three modes, after the mean has been subtracted from the raw data.

The first three spatial EOFs ($E_1 - E_3$) obtained from the beach profiles are displayed in Figure 16.8a. Together they explain about 70% of the variation in the data (the time mean was subtracted before analysis in all data sets). Correspondingly, Figure 16.8b shows the first spatial EOFs ($F_1 - F_3$) for the wave pdfs, which explain more than 96% of the variation.

Table 16.2 Percentage of variance for the first three empirical orthogonal function (EOF) modes

Eigenfunction number	% Variance profiles	% Variance wave pdf
1	-	-
2	34	82
3	25	12
4	11	2
Total	**70**	**96**

The shape functions defined by the EOF analysis can be interpreted as various 'modes' of variation in analogy with Fourier analysis. The first eigenfunction represents the 'best fit' at describing the variation in the data. The second eigenfunction represents the 'best fit' to the deviations of the data from the first eigenfunction, and so on. The number of local maxima and minima in the eigenfunctions increases with the order of the eigenfunction. Thus the third eigenfunction is expected to have a more oscillatory behaviour than either the first or the second.

The first EOF describing the profiles (E_1) reflects the presence of a single bar that receives contribution from areas seaward of it. E_2 characterizes the changes in the bar of the profile, and E_3 may be related to the exchange of material across the profile during major storm events. It is important to take into account that the behaviour in the three EOFs modes after 500 m is reasonably stable. Additionally, the temporal EOFs may be analysed to determine trends of profile changes and oscillatory cycles. The EOFs associated with the wave pdfs (Fig. 16.8b) mainly represent seasonal variations in the wave climate and the effect of severe storms.

While some interpretation of the patterns identified by the EOF has been given here, it should be borne in mind that this is a subjective process. The EOF analysis is an entirely statistical procedure that does not incorporate physical understanding to constrain the fitting procedure. Attempting an interpretation of the physical processes on the basis of EOF results alone is not recommended.

Applying CCA to the two datasets (Figs 16.7a and 16.7b) produced a maximum correlation of 0.49 between $\boldsymbol{U_1}$ and $\boldsymbol{V_1}$ (temporal amplitudes of the first CCA modes) (Fig. 16.9a). Figures 16.9b and 16.9c representing the CCA modes show that material moved between the bar area and the foreshore is associated with an increase in the probability of higher waves in the pdf and vice versa. Erosion will occur in the inshore section due to higher waves and the material will be deposited in the area of the bar, if this bar exists.

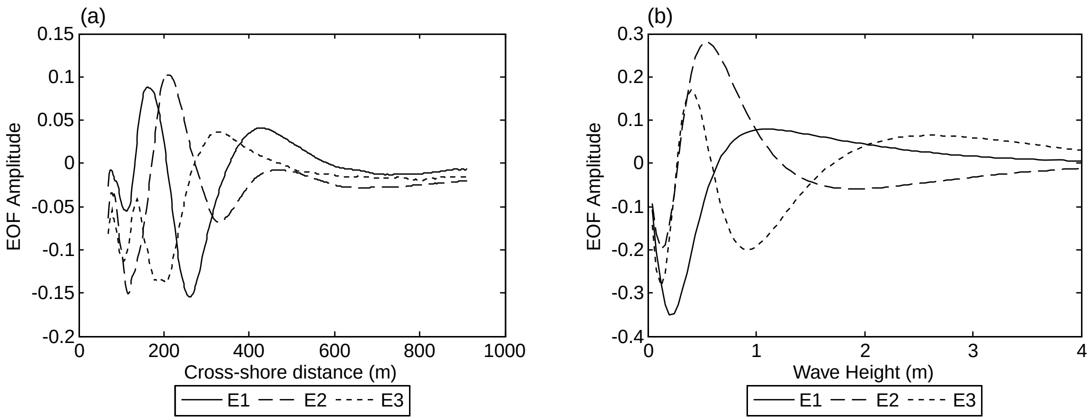

Fig. 16.8 The first three empirical orthogonal functions (EOFs: $E_1 - E_3$) determined from measured (a) beach profiles and (b) probability density functions for nearshore wave height.

Mode H_1 shows the profile elevation that is related with variations in the composite wave pdf as given by G_1. Accordingly, H_1 implies a general decrease across the profile when G_1 causes a decrease or increase in the wave pdf. The higher modes H_2 and H_3 are associated with more complex changes in the wave pdf determined by G_2 and G_3. In order to investigate further the predictive capability of CCA, regression matrices derived from the datasets on profiles and waves up to December 2003 were used to define the correlations between waves and profiles. To make a forecast, measured wave conditions from December 2003 onwards were used together with the correlations to forecast a time series of profiles using the first five CCA modes.

Figure 16.10 shows the measured beach profile and the predicted beach profile made at the profile line 62 for 27 April 2004. Good overall agreement is obtained. The elevation of the bar is underestimated by the prediction and is also shifted slightly towards the shore. A possible cause is that the Rayleigh distribution limits the occurrence of higher waves. That is, in practice there are a greater number of higher waves than predicted by the Rayleigh distribution.

The output from this case is perhaps more of a site-specific insight into how reality deviates from general assumption. A corollary is that such deviations may also be expected at other sites and, if important for the particular design question, the nature of the deviation should be investigated at other sites.

Case Study: Uncertainties in Morphological Models

The central concern of morphodynamics is to determine the evolution of bed levels for hydrodynamic systems such as rivers, estuaries, inlets, bays and other nearshore regions where fluid flows interact with, and induce significant changes to, bed geometry. The goal of this section is to describe some sources of uncertainty within an algorithm for computing bed level change.

Numerical morphological models involve coupling between a hydrodynamic model, which provides a description of the flow field leading to a specification of local sediment transport rates, and an equation for bed level change, which expresses the conservative balance of sediment volume and its continual redistribution with time (see Fig. 16.3).

The majority of the numerical models that have been developed are based on a series of modules that describe the hydrodynamics, sediment transport and bed evolution separately. The

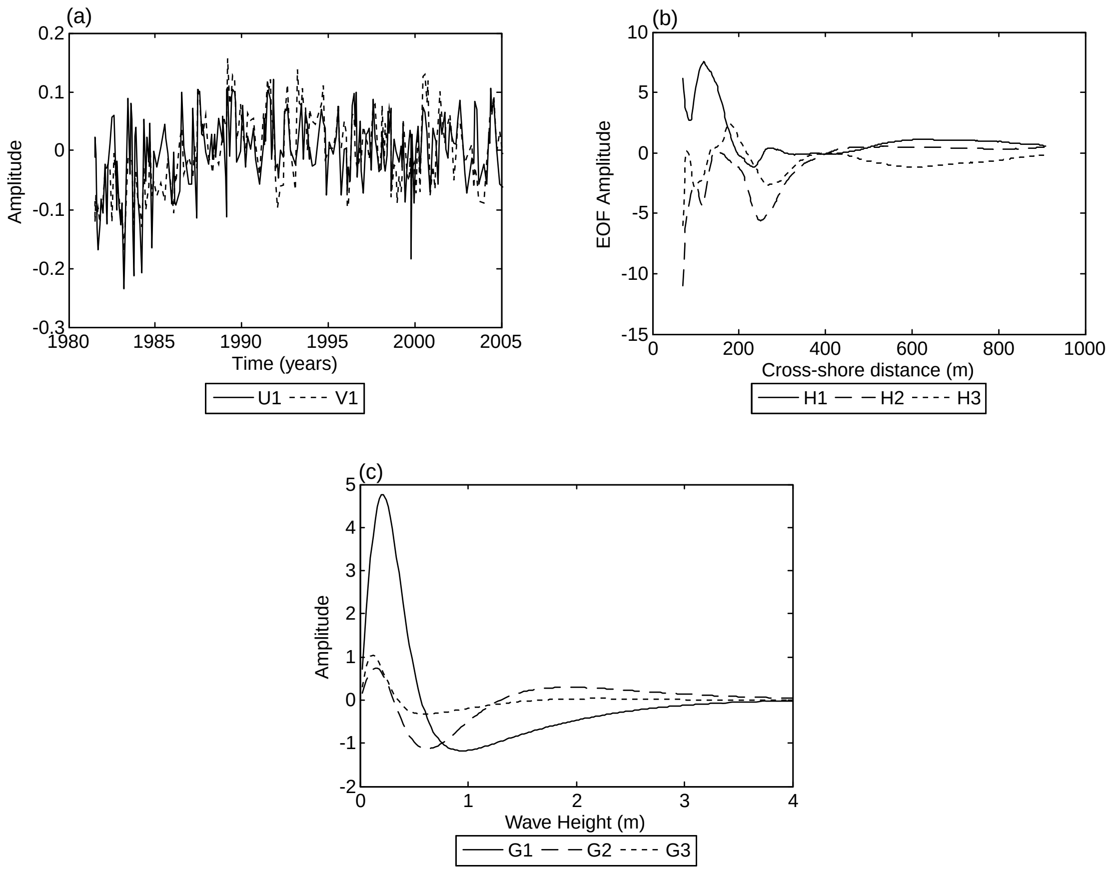

Fig. 16.9 Results of the canonical correlation analysis (CCA) between wave height probability density functions (pdfs) and beach profiles: (a) temporal amplitudes of the first CCA modes (U_1 and V_1); (b) spatial amplitudes of the first three CCA modes for profile elevation ($H_1 - H_3$); and (c) wave height amplitudes of the first three CCA modes for wave height pdf ($G_1 - G_3$).

hydrodynamics is calculated assuming the bed level remains constant. Then the sediment transport is calculated according to the results of the hydrodynamic module. Following on from this the bed level is then updated according to the predictions made by the sediment transport module. Further details of this modular approach can be found in Johnson and Zyserman (2000), Zyserman and Johnson (2002), Karambas and Koutitas (2002) or Pedrozo-Acuña *et al.* (2006).

An important consideration when dealing with hydro-morphodynamic models is the fact that morphological evolution happens on a longer timescale than hydrodynamic evolution. Hydrodynamics can vary greatly over one wave period whereas the morphodynamics requires a greater timescale to evolve. To overcome this problem an appropriate time step needs to be chosen to update the bathymetry; typically this is a longer time step than that of the hydrodynamics. In order to show how uncertainties with regards to this selection of the time step can be avoided, this section introduces a sensitivity test of this parameter and its effects on the induced bed level changes. This is done with results from the morphological model presented in Pedrozo-Acuña *et al.* (2006).

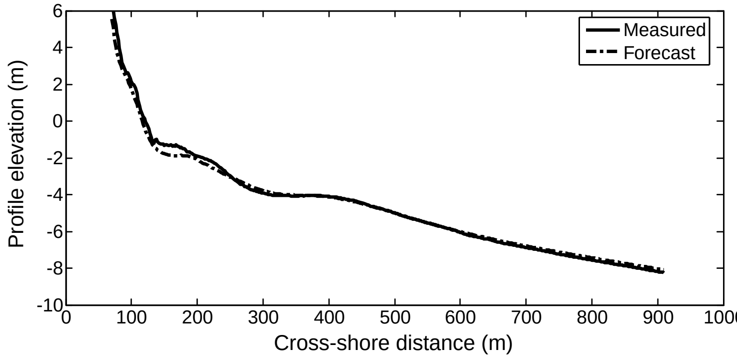

Fig. 16.10 Measured and predicted profiles with Rayleigh wave distribution at the Duck site (date 27/04/2004).

The selected test case is defined in conformity with a gravel beach studied in their Grosse Wellen Kanal (GWK) experiments. That is, we employ a plane beach with a slope of 1:8 with a grain size set to 0.021 m, and the selected waves are defined by a JONSWAP spectrum with $H_s = 0.6$ m, $T_p = 3.22$ s and $H/L = 0.05$. The spatial step defined by $\Delta x = 0.129$ m is the same in all the numerical experiments carried out in this test. The hydrodynamic time step is given by $\Delta t = 0.00076$ s (i.e. instantaneous) and the selected morphodynamical time steps are given by 3 s, 6 s and 12 s. To ensure that the selected morphodynamical time step (Δt) is reliable in the calculation, the variability of the model output with regards to changes in the morphological time step must be small.

Figure 16.11 presents the results for these three experiments, showing the evolved profiles after the action of 100 wave periods. Here the model output treated as a stable solution is shown as a dash-dot line. This figure illustrates the reliability of the model over the selected range of morphodynamical time steps.

In contrast, Figure 16.12 presents two numerical experiments on the same plane slope that show divergence from the stable solution given by 6 s. These morphodynamical time steps correspond to 0.076 s and 100 s. This result illustrates the importance of careful selection of this time step in the model; if these sensitivities are not studied, the uncertainties in the predictions will not be appreciated, with the danger that unreliable output is used subsequently in scheme development.

This case highlights the importance of specialist knowledge in running, and interpreting the output from, numerical models used in the design

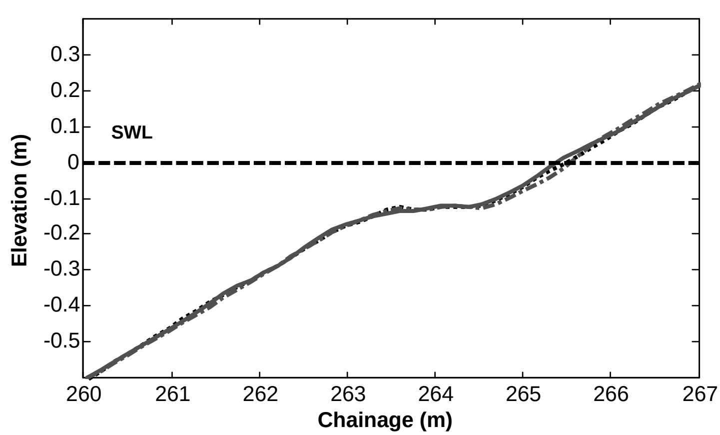

Fig. 16.11 Sensitivity test of induced beach profile changes to selected morphodynamical time step: convergence (– · – ·) 3 s; (——) 6 s; (· · · · · ·) 12 s.

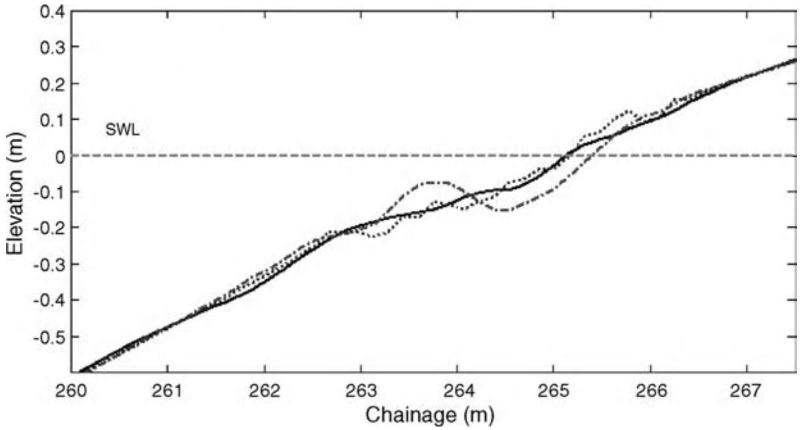

Fig. 16.12 Sensitivity test of induced beach profile changes to selected morphodynamical time step: divergence (· · · · · ·) 0.076 s; (——) 6 s; (–· – ·) 100 s. SWL, still water level.

process. Gaining an understanding of the model results from an analysis of their sensitivity to changes in model parameters gives the engineer a feeling for the robustness of values chosen for any design calculations.

Case Study: Risk Assessment of Cliff Erosion

Successful management of the coast requires a clear understanding of the risks of coastal erosion and instability. This section outlines a risk assessment procedure that was developed for local authority engineers to use for decision support in shoreline management, and which can run on a personal computer spreadsheet program. In this context, decisions have to be made that have practical consequences. Very often, the luxury of waiting many years to compile detailed measurements is not available, information is limited and local knowledge and engineering judgement can provide valuable input. The methodology presented here is based on the source-pathway-receptor risk model introduced by DETR (2000).

Components of the methodology

One single approach is adopted here irrespective of the scale or data. It comprises three basic components, the only difference being the level and accuracy of analysis that sits behind them (Meadowcroft *et al.* 1995; Environment Agency 1996; Hall *et al.* 2003).

As illustrated in Figure 16.13, the components are:

- the cliff instability and erosion process, i.e. the mechanisms and rate at which it might occur (definition of the hazard);
- any resistance to cliff instability and erosion, i.e. coastal protection and slope stability measures that slow erosion (modification of the hazard);
- the location and value of the asset(s) of interest (the risk).

Each component has a number of factors that require consideration and a range of associated techniques, depending upon the degree of sophistication of the analysis.

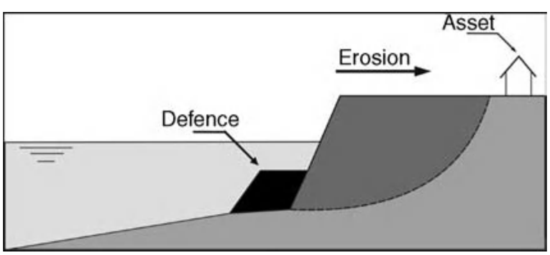

Fig. 16.13 Components of the methodology.

For example, at a well-monitored site it may be appropriate to use a quantitative model to predict cliff erosion rates. Conversely, in a situation where there is little or no recorded information, local knowledge and engineering judgement are necessary.

Irrespective of the simplicity or complexity of the techniques, each should lead to the same basic output: a timeline to cliff erosion and a timeline to defence failure. The principal difference between the outputs from various techniques is the accuracy and the level of confidence.

The general idea is to define, by means of a probabilistic description of failure and erosion, a probability of erosion for a given distance or time, which considers the interaction of both elements, the natural process of cliff erosion, and the presence or not of a coastal structure.

Risk assessment of cliff erosion

With regard to cliff erosion there is uncertainty over when the cliff will erode, by how much and if this would be instantaneous or gradual. With regard to resistance to cliff instability, there is the impact of failure of coastal defences and sea level rise.

The key components to determine the erosion versus time plot are cliff instability and the erosion process. These curves might be generated on the basis of detailed cliff erosion models, on extrapolation of historical erosion rates, or local knowledge and engineering judgement.

Figure 16.14 illustrates an example of the required input, with the black solid line indicating the best assessment, and the degree of uncertainty shown by the dotted and the dashed lines. Hence, in this example, the cliffs are expected to erode inland by an average distance of 102 m over the next 100 years, although there is potential that the cliffs could erode by as little as 100 m, or by as much as 115 m over the next 100 years.

Coastal defence assessment procedure

In a similar manner, the probability of defence failure can be determined. In all cases, taking account of the influence of defences has two components:

- a general deterioration over time, i.e. due to general wear and tear at some point in the future it will cease to be effective;
- a failure of the defence due to design conditions being exceeded, e.g. destroyed by a storm, or undermined by falling beach levels (forcing conditions).

Both of these are variable but in different ways. With regard to deterioration there is uncertainty over the time at which the defence will become ineffective, and indeed if this would be instantaneous or gradual. Failures resulting from environmental conditions can be determined from an annual probability of exceedence.

Factors including climate change and alongshore interactions can also be incorporated into any of the techniques using appropriate

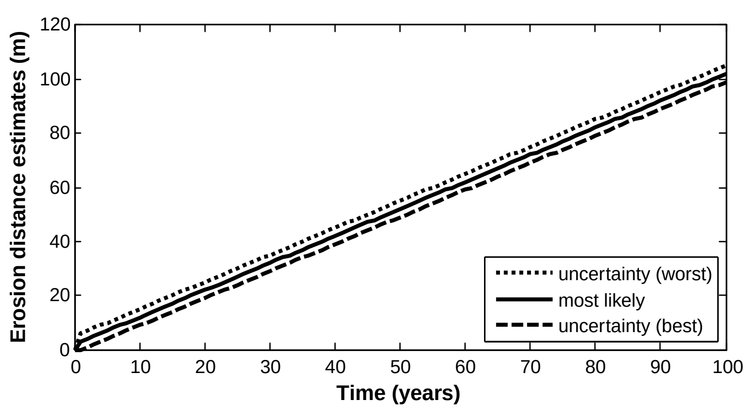

Fig. 16.14 Erosion profiles.

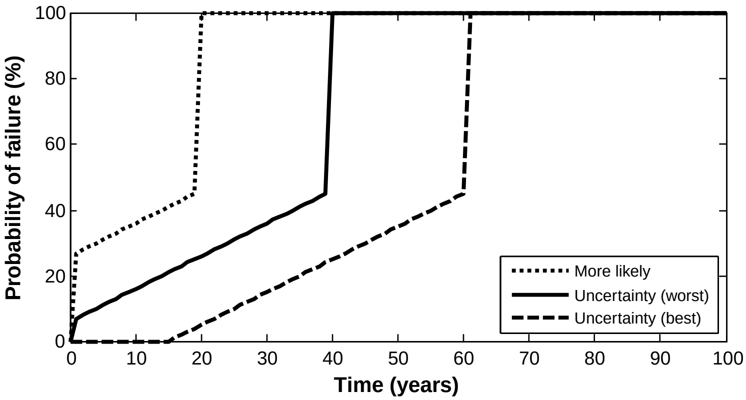

Fig. 16.15 Probability of failure chart.

allowances. Alongshore influences upon defence failure, for example structures restricting the supply of sediment to a frontage, can almost entirely be accommodated through the consideration of foreshore levels. For example, if such restrictions do apply then there is a need to account for that in the foreshore levels or assessment of foreshore volatility.

Figure 16.15 illustrates the required input, with the solid black line indicating the best assessment, and the degree of uncertainty shown by the dotted and dashed lines, which are considered to approximate to 5% and 95% confidence limits:

So, in this example, it is expected that the structure is most likely to be effective for another 40 years, but it might collapse after 20 years, or could last 60 years. During the period leading up to that, the User has assessed that there is a 1% chance of storm conditions exceeding design conditions year-on-year and leading to its failure, although this could be as much as 1.5% or as low as 0.5%.

In other words, under the 'best assessment' the graph is saying that the defence will definitely have failed by year 30, but recognizes that there is a small chance that this failure could actually happen this year, next year, or at any point forward.

Range of techniques for determining recession and probability of failure

The techniques employed to evaluate both the erosion curves and the probability of failure have different levels of complexity. These components can be determined by detailed modelling or empirically from local knowledge and engineering judgement.

Once both curves have been defined they may be combined to obtain probabilities of erosion to a fixed asset or spatial distribution of erosion at a given time in the future. This is done under the definition of three main scenarios as given in the next section.

Definition of scenarios

There are three possible scenarios that can modify the erosion distance estimates. These are defined by the different time estimates of when the failure of the structure is expected, from the probability of failure curve:

Scenario 1: The first scenario is then constructed with the erosion profile curves considering the three time lags defined by the probability of failure. The worst case is represented by using the worst time of failure with the worst erosion profile scenario. The intermediate erosion profile is defined using the best estimate of the time of failure against the best scenario for erosion profile. Finally, the best scenario for the erosion profile is combined with the best estimate of the time for the structure failure.

Scenario 2: The second scenario is constructed from the erosion profile assuming that the erosion line stays in the same position but once erosion takes place there is a catch-up, which would be a straight line up from the zero erosion to meet the erosion profile after a set period of time defined by the User.

Scenario 3: The last scenario considers the effect on the erosion profile if the defence had not been in place for a certain period of time. Thus the erosion profile is shifted to the left to assume this would have been the erosion had the defence not been in place (the User can specify how old the defence currently is). After this process is carried out the sheet works out a catch-up (in a similar way to Scenario 2), to construct the given erosion profile.

The methodology provides the User with the ability to investigate the answers to two particular types of question:

1 What is the probability of erosion for an asset located at a given distance X?

2 What is the probability of erosion for a given time in the future?

Further information on the methodology and spreadsheet can be found in Defra (2006) and Pedrozo-Acuña *et al.* (2008).

This example demonstrates how 'ownership' of some of the uncertainty can be passed back to the user of the technique – thereby allowing engineering experience to be included directly into the process. The visual interface has also proved to be a powerful way to communicate the nature of the risk to non-specialists.

Concluding Remarks

In this chapter some of the uncertainties associated with modelling coastal processes have been discussed. Methods for identifying, quantifying and to some extent mitigating uncertainty in coastal modelling have been illustrated through a set of case studies. These examples cover applications of modelling tidal flows, and statistical and dynamical process prediction of beach profile change as well as cliff erosion.

The different types of model that are available for forecasting have also been discussed. In practical engineering applications other considerations, apart from purely scientific ones, come into play. Thus the quantity and quality of data can be a very important factor in determining the type of model used. In fact, the quality of the data is one of the main sources of uncertainty in modelling. Even given a perfect model, if inaccurate data are used as input, the forecast will be affected. In situations where the available information is of poor quality, a model that gives qualitative forecasts may actually be of more practical assistance in informing a decision, than an inaccurate forecast produced by a sophisticated model.

One promising area of development is ensemble modelling. This involves running models many times over, with slightly different but realistic initial conditions, to create a collection (or ensemble) of possible future outcomes. The ensemble is considered to be a sample of the population of all possible future outcomes. The sample average and variance can be computed directly to provide an indication of the likely outcome and the expected variation about this. This type of approach to handling uncertainty is currently the subject of intense research activity. In the coastal literature there are some examples of this type of approach, using the simpler empirical or hybrid models (Dong and Chen 1999; Reeve and Spivack 2004), and data-driven models (Reeve *et al.* 2008). These can provide useful indications of shoreline behaviour and sensitivity. The development and implementation of an ensemble modelling approach using detailed process models has yet to accomplished, and is the subject of current research.

The concept of balancing risk and benefit is central to engineering. It is rarely up to the engineer alone to decide this balance. Nevertheless, it is often the engineer who has the important role in quantifying, as clearly as possible, the facts that inform financial and political decision-making. Designing flood defence infrastructure in the face of uncertainty has been a primary hallmark of engineering design for many years. Cardinal elements of probability theory have been used extensively but are frequently well camouflaged in what, on the surface, appear to be essentially deterministic approaches to design.

With specific regard to flood and sea defences, the UK's Department for Environment, Food and Rural Affairs (Defra) has promoted a probabilistic approach to their design through the publication of Project Appraisal Guidance Notes. These define procedures that must be followed in applications to the UK government for funding assistance towards flood defence and erosion protection works. Amongst other factors, these procedures require designers to consider carefully the uncertainties inherent in their scheme, and to quantify these as well as possible. Where good information about the construction materials and the loadings is available, a formal probabilistic approach can be adopted, thereby providing a better understanding of the uncertainty, leading to less conservatism in design and thus improving cost efficiency.

Finally, we mention a few words of caution. While reliability theory and probabilistic methods can help to quantify uncertainty they should not be considered a panacea. Indeed, one danger of applying such methods 'off the shelf' is that not all the available information is used. An example of this is when a discrepancy occurs between the results of a reliability analysis and experience. This might arise either because experience has been incorrectly interpreted or because there is an error in the reliability analysis. An examination of the causes of the discrepancy will often lead to a better appreciation of the essence of the design problem, and thereby to an improved design solution.

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Part 6
Policy and Planning

Flood Risk Science and Management
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17 The Practice of Power: Governance and Flood Risk Management

COLIN GREEN

Introduction

Fundamentally, governance is about power: who has it, who should have it, and what forms of power may be used for what purposes by whom. Therefore, definitions of governance are necessarily contested; in particular, any attempt to define 'good' governance specifies both who should have power and what forms of power should be used. This chapter seeks to set out instead to analyse the nature of governance, and the issues that must be addressed, rather than to be prescriptive as to the most appropriate allocation, forms and uses of power. Chapter 18, on stakeholder engagement, then focuses on the problems of governance in a complex society.

Analytically, governance is centred upon dualities as shown in the classic Yin-Yang diagram of two conflicting but complementary elements. Not only must the two elements be managed as a whole, but commonly each element can only be properly understood in conjunction with the other. The Yin-Yang diagram is thus an early example of the 'figure-ground' problem: for example, the Rubin's optical illusion of the drawing of a candlestick that can also be seen as the profiles of two faces (Kennedy 1974). Governance is full of such dualities, notably:

- power – rules
- institutions – organizations
- means – ends
- interests – legitimacy
- technology – governance
- roles – relationships
- rights – duties.

In other words, it is impossible to meaningfully discuss one without the other or to separate the two parts.

Sustainable Development and Governance

Sustainable development requires us to do more with less. Hence, the three obvious questions are:

1 More 'what'?
2 Less 'what'?
3 How?

The more 'what' is usually taken to include the achievement of societal objectives such as development for those currently in poverty and justice (Defra 2005). The less 'what' is taken to mean both the more efficient use and the more sustainable use of natural resources. The 'how' question is the critical one; doing more with less requires that we do better than we have in the past. The extent to which we can do more with less, i.e. to be more successful in achieving our objectives whilst both using less resources and using those resources more sustainably, is determined by:

1 The physical and chemical laws. For example, an engine cannot be more efficient than a Carnot engine nor can we outwit the law of the conservation of mass. These laws define what is theoretically possible.

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2 The current state of technology determines how close we might get to these theoretical limits at any given moment at time.
3 To achieve that technological possibility requires arranging resources in the right place, in the right quantities, at the right time. This is an organizational as well as a spatial problem.

The extent to which one form of organization will enable resources to be used more effectively and sustainably than another form of organization is one aspect of governance. This aspect is the simple functional question of how do we do more with less through the better organization of the conversion of resources and the bringing together of resources. There is evidence that there are considerable frictional costs in transforming resources to meet objectives, including Coase's (1988) 'transaction costs' and Stiglitz's (2008) 'information costs'. As Coase (1991) argued, consideration of the relative transaction costs of different institutional frameworks has to be a central concern. At the simple level, resources can be wasted: labour is wasted if the tools, materials or equipment are not simultaneously available, or the necessary precursor tasks have not been completed.

Additionally, different institutional frameworks have a significant effect upon the efficiency with which resources can be transformed to deliver objectives. One of the arguments for framework agreements with contractors is that in a competitive contract, each contractor has to tender the bill of quantities and not the job (Egan 1998). That is, if the contractor could do the job cheaper than by the means specified in the bill of quantities, then the client cannot take advantage of this knowledge.

Another example is that institutional frameworks can inhibit or prevent the adoption of specific technologies that offer a potential improvement in efficiency. For example, the legal definition of a 'sewer', essentially as a pipe, has been argued to be inhibiting the adoption of Sustainable Urban Drainage options (SUDS) SUDS by the wastewater companies.

One side of governance is therefore: what social relationships are most effective in relation to the use of resources. The other, equally necessary, aspect of governance is: what ought to be the nature of social relationships?

What is Governance?

There have been many attempts to define 'governance' (Paproski 1993; Allison 2002; Evans *et al*. 2005; Moench *et al*. 2003; Rogers and Hall 2003; Cleaver and Franks 2005; Moretto 2005; Swyngedouw 2005; Brandes and Maas 2006), either exclusive of government or inclusive of it. Because governance is about power and who should have it, the definition of governance, particularly of 'good governance', is necessarily contested. But probably the best descriptive definition of governance as a process is:

> The exercise of political, economic and administrative authority in the management of a country's affairs at all levels. Governance comprises the complex mechanisms, processes, and institutions through which citizens and groups articulate their interests, mediate their differences, and exercise their legal rights and obligations.
>
> UNDP (1997)

Governance is therefore about the joint problems of how we decide what to do, and then do it.

Effectively, governance is concerned with power (Lukes 1974):
• Who has it now?
• Who should have it?
• Who may use what forms of power for what purposes?

'Power' here is being used in the purely functional sense of the ability to change or influence the world. The domains of power are therefore over:
• the physical world – the ability to physically change the world;
• the self – the scope that the individual has to act;
• the others – the extent to which one party can influence the actions of others.

Ultimately, all changes involve some physical change in the world, i.e. physical power exercised by some individual or group. Outside this definition,he ability to influence others to act in a certain way is defined as social power.

In this functional sense, power is anything that works; if something can be used to induce change then that is a form of power. Hence, there are many different forms of power (Green et al. 2007); anyone who thinks that because a 2-year-old has neither physical power nor money then that child has no power has obviously never had a toddler. Power thus varies in its range of application and its strength. In this empirical sense, information and skills are forms of power, as is access to reasoning. Science is therefore not neutral because, potentially, it is power.

If 'who has power now' is a pragmatic question, the issue of who should have power raises both questions of effectiveness and of justice. The first is a question of how power should be allocated so as maximize the conversion of resources sustainably into societal objectives. The latter question is clearly central when the issue of power over self is raised and the ability to influence others necessarily impacts upon their power over self. Power over self and the power of others over self are the key ingredients of human rights (Freeman 2002). The question of equality is not therefore a question of being nice to other groups by gender, age or race, or of seeking to promote equality of outcomes, but is centred on the equality of power so that all groups have the same right to make choices, and the same potential range of choices.

To be effective, the range of power must encompass all of that which is to be changed and have sufficient strength to induce the change. A crucial change in flood risk management is the shift from a reliance on physical power – an organization having the capacity to build some structure – to the requirement for social power: the ability to influence the behaviour of others. A second change is that frequently no single organization has an adequate set of powers either to make the change in the physical world or to influence those who must make such a change.

Who Has Power?

Who has what power now determines what can be done and where it can be done. Thus, it is futile to call for the control of development on floodplains if no organization has the power to restrict development (e.g. there is no zoning of any kind in Houston), or if those powers are ineffective -for instance, many developing countries are experiencing rapid informal development, a process that ignores not only planning requirements but also ownership of the land. So, a first concern in relation to power and social justice in relation topower is whether any organization or organizations acting alone, cooperatively or collaboratively, have 'ownership' of the power to deliver a particular technology.

The pragmatic problem of power is that unless some organization has the power to act in a particular way in a particular place then that particular flood risk management intervention cannot be undertaken. But power is bounded; rules create boundaries to power in terms of the types of power that can be used, where, and for what purposes. Thus, a key duality is between rules and power. So, a key function of rules is to delimit one or more of what the subject of the rule **must not do**, **may do**, and **must do**. Consequently, rules define functional boundaries as well as geographical boundaries.

In turn, the standard definition of an institution is that it is created by the existence of a formal or informal set of rules (Uphoff 1986; North 1990; Scott 1995) where rules can govern both its internal behaviour and its external relationships. Rules create and delimit power, whilst a very effective form of power is the ability to define the rules; so rules both create and limit power, whilst being created by power. Legal frameworks are the archetypal system of formal rules, whilst governments are the archetypal setters of formal rules.

Institutions have to be distinguished from organizations; any organization is also an institution because an organization is governed by both internal and external rules. But an institution does not necessarily result in an organization. Similarly, the different academic disciplines are clearly institutions, ones defined by informal rules and, in some cases, also by formal rules (such as the legal definition of a lawyer).

Thus, practical governance is limited by the systems of rules, which define powers and the boundaries created by rules. Common problems with those systems of rules are:

1 fragmentation
2 gaps
3 ambiguities
4 overlaps.

Fragmentation of responsibilities is inevitable because the physical and social worlds have to be broken down into manageable pieces. In many countries, the constitution restricts the extent to which the form and nature of those administrative organizations can be changed. This is notably the case in France, Germany and the USA. In those circumstances, there is no option but to seek ways of collaborating across boundaries. Conversely, in the absence of such a constitution, the UK has progressively centralized administration, but the effect of such centralization is simply to internalize the problem of working across boundaries, those within the organization. Hence, all important water management problems are transboundary problems; it is only the nature of the boundaries that differs.

In particular, flooding can be summarized as being the result of weather, antecedent weather, and landform as modified by land use. As only the last can be modified in the small scale by human action, flood risk management is fundamentally about spatial planning. Hence, a key form of cross-boundary working is between water management planning and spatial planning.

Equally, there are often gaps in responsibility, particularly where the organization form has not changed to reflect the new understanding of the system. The organizational form often also embodies a technological approach to the problem. This is particularly true where the traditional approach involved the use of physical power; for example, the construction of embankments and reservoirs. The new approach of sustainable water management (Defra 2008) instead relies heavily upon social power: the ability to influence others through flood warnings, flood resilience or resistance, or to influence spatial planning policy. This shift is revealing gaps in responsibility that were previously concealed. Moreover, the form of water privatization adopted created arbitrary boundaries.

Such gaps are often accompanied by ambiguities: a lack of clarity as to who has what power to do what where. For example, Sheail (2002) notes that until some 80 different bodies were replaced by the Great Ouse Drainage Board, it was impossible to determine who had responsibility for remedial works on the tidal River Ouse. Overlaps create ambiguity; for example, the ambiguity between the duty of the riparian owner under the Land Drainage Act 1991 to maintain a watercourse and the permissive power of the relevant Internal Drainage Board to carry out maintenance works for land drainage purposes.

Therefore, a starting point in any practical exercise in governance is to prepare an institutional map (Green *et al.* 2007), combining spatial boundaries and functional boundaries. Definitions are frequently central to defining the functional boundaries: most obviously, what is a 'river' and what is a 'flood'? In England, unless otherwise defined in the particular case, the adjacent landowners own the land under the river. In other countries, the river and often some land on either side of the river, is state property – an issue when rivers tend to move. If a flood is too much water in the wrong place at the wrong time, such events occur from a multiplicity of reasons. Consequently, if insurance does or does not cover 'flooding', it is necessary to define exactly what kind of events are covered and what are not otherwise an exclusion of 'flooding' will cover flooding from a burst pipe either indoors or externally. Similarly, Hurricane Katrina has been followed by many arguments between the insurers and the insured as to whether damage to property was caused by wind damage, covered by the standard insurance property policy, or flood damage, which is not covered by those policies.

Figure 17.1 sets out such a functional institutional map of land drainage in England. The basic framework is a schematic outline of the main physical components of the drainage system,

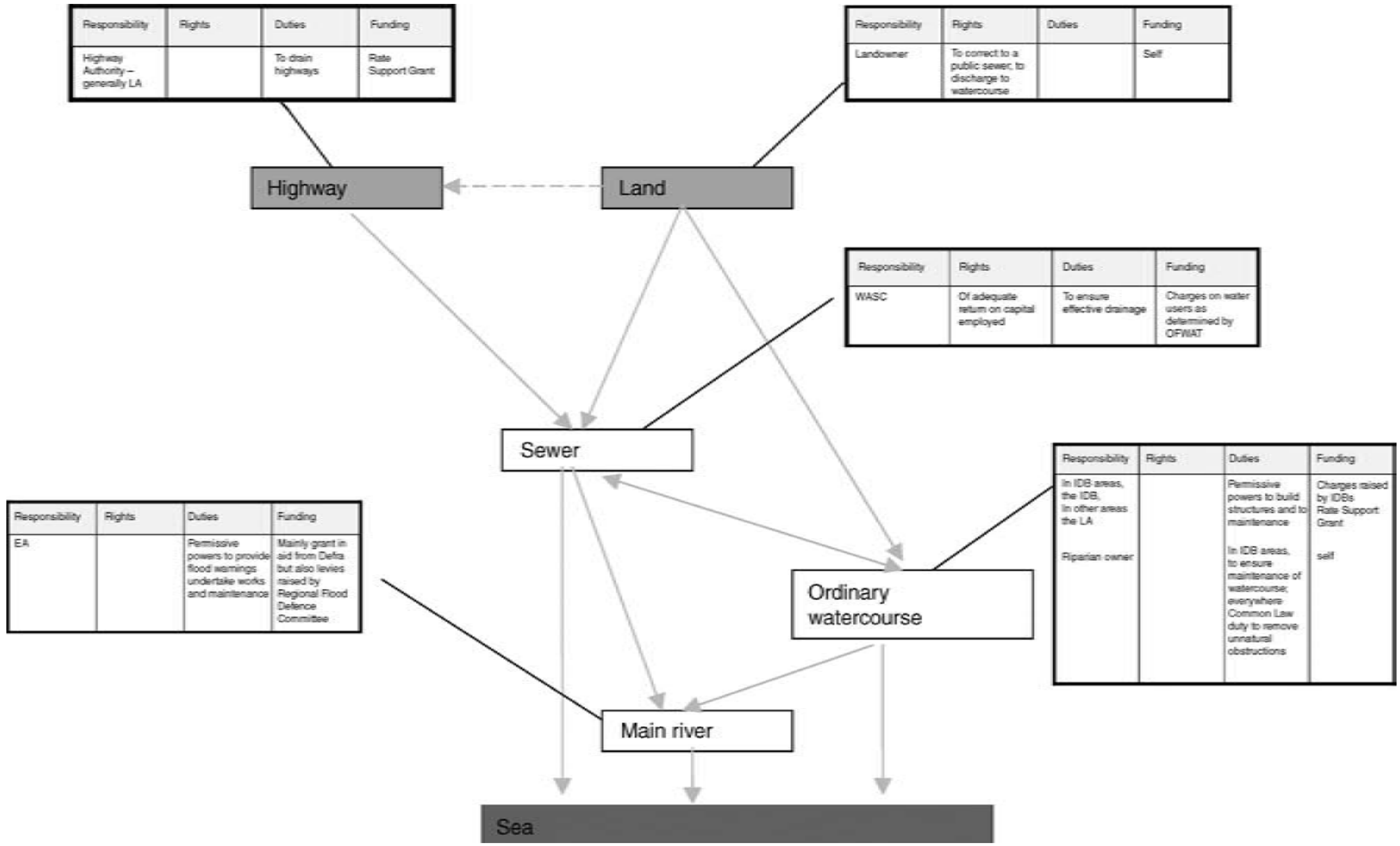

Fig. 17.1 Institutional map of land drainage in England. EA, Environment Agency; IDB, internal drainage board; LA, local authority; OFWAT, Water Services Regulation Authority; RFGC = Regional Flood Defence Committee; RSG = Rate Support Grant; WASC, water and sewerage company.

which include some rule-defined elements in the form of 'sewers', 'ordinary' and 'main' watercourses. Some ambiguities are already apparent: whilst the individual Highway Authorities are responsible for draining the roads, land is sometimes drained to the highway. Again, in some cases, when a watercourse was culverted over it was designated as a Public Sewer, but then reverts to a watercourse when it returns to an open channel. If the watercourse was not designated as a Public Sewer, it remains a watercourse.

Attached to each component are the parties with responsibility for it; the parties each have:

- duties, what they must do – the power over them;
- rights, the power to act – what they may do;
- source of funding – a significant form of power.

What can be seen from Figure 17.1 is the absence of any duties on landowners with respect to draining their land so that a problem such as the so-called 'muddy floods' from agricultural land escapes the legislative framework (Boardman *et al.* 2003). Overall, no organization has the responsibility for ensuring that land is effectively drained, but instead specific organizations have responsibilities for specific physical assets: the Water And Sewerage Companies (WASCs) have responsibility for anything that is designated a 'Public Sewer'. Or, organizations have responsibility for parts of the system: notably the Internal Drainage Boards (IDBs) are responsible for ordinary watercourses in an area where an IDB exists, with the local authority having the responsibility in those areas where there is not an IDB. The duties of the landowner adjacent to a river – the riparian owner – are similarly confusing: there are duties under common law (Howarth 1992) to remove unnatural obstructions, and under statute law, the 1991 Land Drainage Act, there are unspecified maintenance duties to the watercourse. Specifically, the Land Drainage Act does not define to what standard a watercourse should be maintained.

Of particular concern is the lack of definition of the boundaries between responsibilities; so, for example, currently the WASC cannot refuse to connect a properly constructed drain to its public sewer. Equally, whilst the riparian owners have a responsibility to remove unnatural obstructions, there is no control of discharge of unpolluted water to a watercourse. In principle, every right or entitlement should be accompanied by a corresponding duty or obligation on the receiving party. In many cases, one side of the equation is implicit rather than explicit. Thus, the Royal Commission on Land Drainage (1927) referred to upland owners considering that they had a right to drain their land and consequently imposed heavy costs on lowland landowners in order to protect themselves from the resulting flooding. The same issues that concerned the Royal Commission replayed 80 years later in the Pitt Report (Pitt Review 2008).

The Differentiation of Interests and Power

The technological and economic development of societies has resulted in a diffusion of power. In early societies, power rested largely in the monarch, the priests and large landowners. Essentially, everyone else was a subsistence farmer who produced relatively small surpluses of food and was largely self-sufficient in clothing, building and utensils.

Technological development has been accompanied by specialization of skills and tasks, and has resulted in the greater complexity of social organization, and hence greater exchange of goods and services between members of that society. That differentiation necessarily resulted in differentiation of interest – the interests of one group were increasingly likely not to be those of another. A market economy in particular depends upon competition, or conflict, between potential suppliers; equally, it sets up an adversarial relationship between consumers and suppliers.

All those sharing an interest realized that to have a share of power it is necessary to organize, with the medieval guilds (Postan 1972) being early examples. Those occupations that claimed to be a profession or trade thus organized early but by the early 19th century so were the working classes, both in terms of trade unions but also in the different forms of mutual societies (life insurance, retailing, health insurance and so forth). These became widespread so that by the end of the 19th century at least a third of the working population was a member of some mutual society (Gorsky 1998).

These were organizations to promote the self-interest of their members, but increasingly there emerged mass movements to promote the interests of others. Perhaps the earliest in the UK was the Anti-Slavery movement (Colley 1992), and the 19th century was full of non-governmental organizations (NGOs) seeking improvements in conditions (Walvin 1987), be these the extension of voting rights (e.g. the Chartists); the extension of education; the sanitarian reforms (notably Chadwick) seeking improvements in public health and living conditions; the Settlement Movement; the Allotment movement; and the improvement of working conditions, particularly for children and women. These were followed by more heritage-based NGOs such as the National Trust. Now, NGOs might be broadly classified as those concerned with human rights, or the alleviation of poverty or environmental issues. Whilst the concept of 'community' has eluded any more functional definition than its recognition as a symbolic system to define who is included and who is excluded (Cohen 1985), such communities of interest are as important as the traditional communities of place.

All these different groups are broadly classed as 'civic society' (Seckinelgin 2002); what all share is a single, if often broad, interest. This makes constructing collective action arguably more difficult. It also raises questions of how much weight should given to each interest if this is to be more than a reflection of the current amounts of relative power that each can muster. Again, whilst communities of self-interest have a clear line of accountability to their members, the accountability of NGOs, which set out to promote the interest of others or other species or wider abstract concepts, is more difficult to establish (Lloyd 2005). In turn, legitimacy in a democracy is established by the participation, usually through voting, of the members of that body. This is obviously not possible when an organization seeks to represent an abstract or non-human interest. In consequence, as is developed in Chapter 18 on stakeholder engagement, deciding who can claim to be a stakeholder in the decision, and what powers they can be allowed to exert to influence the outcome, raises moral questions centred on justice.

What Must Governance Do?

Functionally, governance must match the task to the context in which that task is set. There are three key aspects to the task: to match the overall context (itself a synthesis of social, cultural and physical components) secondly, since one function of governance is to allow the effective use of a technology, governance and technology must be compatible; and finally the governance process must be sensitive to the decisions that are needed (or, the reasons why choices have to be made).

Context

The form of governance must match the context in which it is set and from which it emerges. The nature of the water management problem, including the flood risk aspect, varies dramatically from country to country, and region to region. In particular, rainfall varies both over the year and between years, in total amount, and in intensity. Similarly, floods are only one part of the water cycle and should not be managed in isolation lest that management worsen other aspects such as water resource availability.

In England, the density of population and the intensity of land use are both high; flood risk management interventions require both space and place and have to be fitted in within the existing development fabric. Flood risk management must also take account of the development process as a whole; the largest mass migration in human history is taking place in the developing world, with urbanization occurring far faster than it did in Europe or North America.

Cultural factors are also critical. In the UK, traditionally, the stated objective in public policy has been to determine the national or public interest. Conversely, in much of continental Europe, the aim is social solidarity, whilst in China and some parts of Africa, great emphasis is placed upon social harmony (Consedine 1999). As might be expected, those cultural differences are expressed in the prevailing systems of rules in those countries, notably in their legal frameworks. A functional question arises: is any one of these systems more effective at delivering all aspects of sustainable development?

Technology

A continuing question is whether invention is exogenous to culture or society, or whether it is endogenous. That is, whether invention is an almost random consequence of curiosity or whether the area of invention is chosen. What is clearer is that the adoption of invention, innovation, and the likelihood of the successful and rapid adoption of that innovation depends upon the culture and the governance. This was the argument for non-structural (White 1964) flood risk management interventions, the expectation that traditional engineering bureaucracies would not adopt technologies that neither fitted in with their view of the world and their mission, nor were consistent with engineering practices. Equally, environmental groups such as WWF are almost compelled to insist on such intervention strategies as wetlands (WWF Scotland 2007) even when the evidence for their effectiveness is limited (O'Connell *et al.* 2007). Thus, governance and technology are reflective of each other.

Choice

Governance has to address the reasons that make choices necessary. For a choice to exist, there have to be at least two mutually exclusive options, at least one reason to prefer one option to all others, and at least one other reason to prefer one of the alternatives (Green 2003). If all can agree that one option should be preferred to all others then the choice has been made. Hence, the two necessary conditions for the existence of a choice are:

1 conflict, and
2 uncertainty – doubt as to which course of action should be adopted.

Consequently, governance has to be a process that addresses the particular nature of the conflicts, including the conflicts of interest, that make the particular choice necessary and seeks to create confidence that one option should be adopted. It has to resolve those conflicts and it needs to do so in a way that both addresses societal objectives and increases the likelihood that future conflicts will be resolved successfully. It cannot afford to address any single conflict in a way that makes the resolution of future conflicts more difficult and reduces the likelihood of future co-operation or collaboration. The most challenging of those conflicts are those between the different interests of the different groups making up society, and between the different societal objectives brought to the choice.

Governance and Technology

Social relations are central to the 'how' question: what is the best organizational means of bringing the resources together? There are five possible ways of bringing the resources together:

- **Individual:** the individual household or organization is able to mobilize sufficient resources on its own.
- **Competitive:** where the individual household or organization is unable to take action itself, it buys in that capacity from others.
- **Cooperative:** where individuals or groups work together to achieve a single common goal in the short term.
- **Collaborative:** where individuals or groups work together to take actions to achieve what they believe in the long run will be beneficial to all.
- **Coordination by hierarchy:** the 'king-who-commands' model.

If individuals lack the specific powers, such as skill or physical resources, to undertake action on their own, then if they have other power, notably money, they may be able to buy in those resources or capacity from a competitive market. A competitive market offers the capacity to exchange one form of power – money – for physical forms of power. Both economic (Kahn 1993) and legal analyses of markets (Poole 2008) therefore require that the exchange be voluntary, and that power is not used inappropriately. This precludes the use of competition on its own when these conditions cannot be met.

But, these voluntary and non-competitive models be inadequate for one of three reasons:

1 economies of scale may make collective action more efficient;

2 private action may not be possible;

3 the costs of information (Stiglitz 2008) and transaction costs (Coase 1991) mean that an alternative is more efficient.

In these cases, the remaining more 'top-down' options are appropriate. But historically, the cooperative approach has dominated in flood risk management, as in most water management, in the form of Water User Associations. Much of the land drainage in Western Europe was carried out by such cooperative groups, of which the Waterschappen in The Netherlands are simply the best known (Wagret 1967). The traditional municipality is a good example of the collaborative model where a group of people come together and undertake several different activities together rather than a single function. The historic municipalities were of this form and it was these municipalities that very largely delivered water and sanitation to the urban areas of Europe in the 19th and early 20th centuries (Hietala 1987).

It is necessary to distinguish the collaborative model from the coordination model and to avoid describing both as a form of 'state' delivery system. What is a 'state' is itself a matter of contention (Dunleavy and O'Leary 1987). What is notable about states is not their power, but the absence of power and the struggle for legitimacy against other interests. Hence, until comparatively recently, the state was not generally an important player in flood risk management except where the state had the ability to organize corvée labour. Only from the 19th century did it generally have enough money to be able to take action on its own (Hartwell 1981).

An issue in flood risk management is therefore which approach is most appropriate to a particular flood risk management intervention. What is the most appropriate form of intervention is then dependent upon such factors as resource requirements, effectiveness, and the contribution of the intervention to wider societal objectives. In this, one of the key characteristics of flood risk management interventions, as with other aspects of water management, is the physical economies of scale. Physically bigger does generally mean a lower per unit cost. So, building a flood embankment will, beyond some intensity of development, mean that it is cheaper than the individual property owners floodproofing their individual properties (Green *et al.* 2000).

Taking the intervention strategy and the form of governance required to deliver it, which is the best combination? Table 17.1 relates the main clusters of intervention strategy to the forms of governance that can deliver them. The shaded area represents those cases where the intervention is delivered by some user and hence the organizations responsible for flood risk management have

Table 17.1 Main clusters of intervention strategy corresponding to the forms of governance that can deliver them. Shaded areas represent those cases where the intervention is delivered by a user, and hence flood risk management organizations have to exert social power to influence the behaviour of those users

Governance	Runoff control		Delaying/reducing peak flows		Separation of floods and activities					Contingent action	Recovery
	Runoff reduction	**Slowing runoff**	**Offline storage**	**Online storage**	**Dikes**	**Channel improvements**	**Land use control**	**Flood proofing**	**Relocation**	**Flood warnings/**	**Insurance**
Competition								•			•
Cooperation			•	•	•	•					
Collaboration	•	•	•	•	•	•	•		•	•	
Coordination	•	•	•	•	•	•	•		•	•	•

to rely upon social power to influence the behaviour of those users. It does not appear possible for some forms of intervention to be delivered by all forms of governance relationship. In this analysis, three conclusions emerge:

1 That delivering many of the interventions depends upon social power – the ability to influence the behaviour of the end users.

2 The lack of a primary role for competition. That is to say, it is difficult for a profit-seeking company to build, operate and maintain a dike system without breaching the rules of competition and giving a tax-raising power to a profit-seeking company. Competition has a potential secondary role; for example, in the construction, operation and maintenance of interventions undertaken cooperatively, collaboratively or through coordination.

3 Insurance is only possible in the form of some public-private partnership where the state bears a significant fraction of the risk (CEA 2005; Korczyk 2005). In most countries, the state acts as the reinsurer of last resource, paying for all those losses that the commercial retail insurer is unable to transfer to the commercial reinsurance market (GAO 2005). Examples of such systems include flood insurance in the USA, where the federal government carries all the risk (Insurance Information Institute 2005), and catastrophe insurance in France, where the French national government carries the risk above the amount accumulated in the catastrophe fund (de Marcellis-Warin and Michel-Kerjan 2001). A similar system is found in Spain (GAO 2005).

Governance in Practice

So far, it has been argued that the two key requirements of governance are an ability to collaborate across organizational boundaries and the effective use of social power. The assumption has been made that integrated or holistic management is necessary for the delivery of sustainable water management. This assumption needs to be examined more closely.

Integration

Conventional wisdom is that water must be managed in an integrated way; hence, the concepts of Integrated Water Resource Management (GWP 2000) and Integrated Flood Management (Technical Support Unit 2003). There is an immediate contradiction between this desire for integration and the simultaneous call for stakeholder engagement and subsidiarity. Equally, it is necessary to ask what are the potential benefits from integration, and what forms of integration are therefore most likely to deliver those benefits. The argument for integration is that we are seeking to manage a system, one where there are important interdependencies between the elements, and where we must manage the dynamic response of that system. That is, action at any point at any time will have consequences elsewhere in the system. These are the economists' externalities, and a characteristic of flood risk management in the past is that we have simply moved the flood around; by intervening at one point to reduce the risk of flooding, we have simply shifted that risk elsewhere. Secondly, we hope to capture the economies of scale and scope that are frequently possible in water management.

Ideally, therefore, we would like to have a series of independent systems, each of which could be managed in isolation. The obvious problem is that in a modern society there is a high degree of interdependence between the parts. Hence, we are likely to find that we have to choose what is most important to integrate – the achievement of one form of integration requires that we sacrifice another form of integration.

Collaboration

Old style water management involved setting up coordinating agencies, in the form of river basin commissions (Adams 1992). These were frequently scientific bureaucracies. The first problem with this approach was that they simply internalized the boundary problems rather than resolved them; secondly, they sought to deal with the integration problem by accruing more powers.

Thirdly, they tended to treat problems as technical ones that could be solved optimally within their own disciplinary domains. Hence, the not infrequent complaints that they had developed the 'optimum' solution but that this had been stopped by 'politics' – the conflicts and interests that they had failed to take into account in developing their technically 'optimum' solution. Those bureaucracies also were slow to catch up with the current emphasis on stakeholder engagement – the recognition of the diffusion of interests and power.

If collaboration is now essential, there are three questions:

1 Why may collaboration result in more desirable outcomes than independent action?

2 What incentives are there for organizations to collaborate?

3 How can collaboration be successfully delivered?

It is possible to show that under some circumstances, collaboration may result in more desirable outcomes for the organization involved than independent action (Green 2008, 2009). But the incentives have to exist for the organizations to collaborate, and that requires some coordinating action. Similar incentives have to exist for members of those organizations to collaborate with members of other organizations.

But the really difficult bit is to deliver collaboration in practice. Collaboration is conducted by people through the medium of symbolic systems, notably language. Hence, it is the social skills associated with conflict resolution, the management of uncertainty, and the use of symbolic systems for interpersonal communication that are critical. What we require are teachable skills in social relationships and use of symbolic systems for communication. Taken together, those skills are what is often called 'conversation'. That the different disciplines are different cultures adds to those complexities.

Where collaboration has been successful (Imperial and Hennessey 2000; Lowry 2003; Hooper 2005; Moss and Monstadt 2008), the sort of words used to describe the necessary conditions are those like 'trust' and 'confidence'. Technical competence is a necessary precondition for confidence but trust and confidence are more concerned with honesty and truthfulness, and understanding of the motivations of the others (Kohn 2008).

Doing 'Better'

Sustainable development requires continuing change – constant improvement upon what we have done in the past. Thus, it requires both learning and innovation, where learning is commonly defined as more successful adaptation. Both aspects create problems, as does change itself - since that will differentially impact on interests and hence be opposed by those interests. In turn, that change may result in a change in power relationships (Olson 1982). Change is destabilizing and requires different skills to the traditional emphasis on stability; for example, 'learning organizations' have different characteristics from conventional organizations (Hitt 1995; Argyris and Schon 1996).

It is also inevitable that some innovations will not be successful, and therefore what is required are more successful failures –: failures that provide lessons as to how to be more successful in the future. Arguably, there is more to be learnt from a failure than a success. Hence, we have to demand of our institutions that they have more failures whereas conventionally we criticize institutions for failures. At the same time, a strategy for innovation must also ensure that failures are not potentially catastrophic.

Conclusions

In order to become 'better' at governance, we must first understand it and determine what we mean by 'better'. Governance, however, is composed of dualities, including that between technology and governance itself, and is about resolving conflicts. Moreover, it is centred on power and the questions of who has it and who should have it. Consequently, whilst there is a functional aspect of governance, governance also requires we consider the question "...what are social relationships, and what should those relationships be?" to deliver effective flood risk management This is because we now recognise that we have to become *better* at social relationships in order for governance in relation to flood risk management to be effective; these issues are taken up in Chapter 18.

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18 Stakeholder Engagement in Flood Risk Management

COLIN GREEN AND EDMUND C. PENNING-ROWSELL

Introduction

Governance, it was argued in Chapter 17, is about power: who has it, who should have it, and how it may be used (Lukes 1974). It is about how decisions are made and how they are implemented. Thus, it is simultaneously about how to achieve objectives and how to make the best use of available resources in delivering on those objectives. It has to address the reasons why decisions are necessary in the first place, and hence the different forms of conflict. It has to do so in a context where interests are many and power is diffuse so that there are many different stakeholders relevant to a decision, and when social power - the ability to influence others - is increasingly as important as physical power – the ability to change the world. It has to do so when a cooperative solution may offer at least one interest group no advantages over an alternative, and when the potential gains from collaboration lie only in the longer term.

Because governance involves the possession and use of power, it raises all the difficult questions of justice, legitimacy and accountability – different forms of social relationships. For governance to be effective, it is also necessary that we become very good at managing social relations; social power is a skill rather than a property. In turn, those social relationships are managed through different symbolic systems, notably language, and we have to become effective in the use of those symbolic systems, including understanding how those symbolic systems work.

Power is one form of social relationship; the adoption of the governance model also opens up and requires the reconsideration of many different forms of explicit and implicit social relationships, and the reciprocals of relationships, social roles. For example, until recently the role of the engineer was understood to be to determine what society needed, determine the best means of satisfying those needs, and to construct (literally) that best option. Thus, engineers could talk about optimization. Stakeholder involvement has changed all this.

'Stakeholder engagement' is then the practice of social relationships in the pursuit of governance. For stakeholder engagement to be useful and successful, it is necessary:

- For there to be a reason for it: that in some sense, decisions and their implementation will consequently be 'better' than they would otherwise be. In Chapter 17, the argument was developed that, in some circumstances, cooperative or collaborative approaches can sometimes deliver better outcomes than individual action.
- That there is a theoretical basis by which such a better outcome can be achieved.
- That the techniques and skills deliver on theoretical gain. It would be useless if, whilst some 'better' outcome could hypothetically be delivered through stakeholder engagement, there was no means of reaching that outcome in practice.

There are many descriptive manuals available (UNDP 1997, Acland 1990; Creighton *et al*. 1991;

Flood Risk Science and Management
Edited by Gareth Pender and Hazel Faulkner
 ISBN: 978-1-405-18657-5

Weisbord and Janoff 1995; Palmer and Roberts 1998) which cover techniques that can be used in the context of stakeholder engagement; there is much less material available as to the theoretical basis for expecting a better outcome to be achievable and/or as to how it might be achieved.

What is Stakeholder Engagement?

Functionally, the purpose of stakeholder engagement is to achieve a better outcome in regard to a particular choice than would otherwise occur; over the long term, the purpose is to preserve and enhance the capacity to deliver the potential gains from cooperation and collaboration; what has become known as 'social capital' (Bourdieu 1980; Coleman 1988).

Since a condition for the existence of choice is conflict (Green 2003), stakeholder engagement is about the resolution of conflict (Shamir 2002). The pragmatic questions are consequently:

1 Is a resolution of those conflicts possible?
2 What mechanisms and tools will enable such a resolution to be achieved?

The two opposing answers that are heard to the first question are:

1 getting people together will necessarily result in a consensus; or
2 it will only exacerbate conflict and delay decisions.

Neither of these answers is very helpful: both assume the process will inherently result in a particular outcome. Instead, there is a need to focus attention upon the original questions. Equally, it needs to be recognized that the process is difficult, and that perhaps resolving real conflicts of interest ought always to be difficult, and certainly that those whose interests conflict need to be shown respect. Again, simply focusing upon the outcome, that there is something that can be claimed to be an agreed outcome, must not distract attention from the process. If that outcome results simply from creating a new arena of power for those who already have power, or giving power to those who ought not to have power, then the outcome is the result of a bad process.

In Chapter 17 it was argued that in some circumstances either a cooperative approach to a single choice or collaboration over a series of choices would be beneficial to the parties involved. The two possible outcomes of a single decision process are:

1 only winner/loser options are possible, or
2 a win-win option exists.

Whilst the second class, cooperative solutions, are reasonably common in water management, we should expect that many choices will involve the former. The hope here is that by chaining choices together, a collaborative approach over the long run will result in a win-win outcome. Hence, the critical purpose of stakeholder engagement is to build and maintain the conditions under which collaborative approaches will be achieved. This requires that the individual interests accept a loss in one choice in the expectation that they will make gains in consequent choices. What stakeholder engagement can do, therefore, is to widen the negotiating space for the individual stakeholder from a focus on short-term narrow self-interest, of the kind usually ascribed to the rational economic person (Frank 2006), to the possibility of trading off this interest against either long-term narrow self-interest or for some wider interest.

Who is a Stakeholder?

The two starting groups of stakeholders are those who have power to implement action or to obstruct it, and those who should have power. In practical terms, the concern is with two conditions:

1 Between those who have power now. Chapter 17 asserted that nearly all practical problems in water management are transboundary in nature; that is, it is only the nature of the boundaries that differ. Here the necessary power to implement a specific option is fragmented between different organizations, and so only cooperation or collaboration between those organizations can deliver the option.
2 The transfer of some powers to those whom it is considered ought to have a share in power.

Because the aim is to deliver integration, but this has to be done through a fragmented system of organizations, a key organizational skill is increasingly the ability to influence the decisions and practices of other organizations, and the public. Obvious examples are those influencing land use planners to promote and require land use development that, at a minimum, does not increase the risk of flooding and, ideally, reduces that risk. Similarly, the whole purpose of flood warning is to change the behaviour of the public. Therefore, skill at stakeholder engagement is increasingly a key skill for organizations (Le Quesne and Green 2005). The descriptive questions are: who has power now to influence either the decision or its execution? what are those powers and how are they exercised? Such an analysis is the purpose of institutional mapping (Green *et al.* 2007).

The second form of stakeholder engagement is centred upon who ought to have power. The difference between consultation and stakeholder engagement is one of power – who has the power to decide what to do? In public consultation, the power to decide is retained by the organization originally responsible for deciding; under stakeholder engagement, that power is transferred to the stakeholders in whole or in part. Since power is a zero-sum game, if one party gains more power, another has less power. The normative questions are: who ought to have power? what power should they exercise and how?

Since stakeholder engagement is about a share in power, a claim to be a stakeholder is a moral claim to power, a claim that must be supported by a rationale that justifies that claim relative to the claims of others. The three basic questions that have to be addressed are:

1 What are the interests that result in a real claim to power?

2 What forms of and how much power should one interest have relative to the claims of other interests?

3 How can those powers be properly exercised?

The moral question of who ought to have power cannot be avoided. Thus, President Museveni (2002) stated: 'Therefore, the arrogant so-called Non Governmental Organizations [NGOs] that interfere with the construction of hydro dams in Uganda are the real enemies of the environment'. The implied question is: what is the basis for the claims of those NGOs to have this power? Neither of the answers that 'we know best' or 'because we have the power to block loans' are morally attractive. A desire to be a stakeholder should not necessarily translate into the right to be one.

So, who can claim to be a stakeholder? Any definition is a duality: it simultaneously distinguishes between what is contained in that definition and what is excluded. Thus, the definition of a stakeholder given, for example, in the Common Implementation Strategy for the Water Framework Directive (2002), is an invitation rather than a definition: it potentially excludes no one. The definition of a stakeholder and form of power that they may possess can be treated together. Power comes in a number of different levels such as:

- the right to information;
- the right to be heard;
- the right to have those interests properly considered;
- the right to take part in the decision process;
- the right to co-determine the decision.

Thus, different interests may legitimately be accorded different levels of power: the question becomes one of who has a moral claim to what level of power. The Aarhus Convention (UNECE 1998) establishes the first three powers as rights to the public where the 'public' is defined in Article 2 as: 'natural or legal persons and, in accordance with national legislation or practice, their associations, organisations, and groups'. The arguments over the proposed EU Directive on environmental justice have in turn been about what associations, organizations or groups should be recognized as having standing. This debate is not only a struggle for power but also about legitimacy and accountability.

It is the latter two levels of power that create the difficulty. The starting point of that claim is that those making it have an interest; those interests vary and any single individual or organization may have conflicting interests. A reasonably generic model of the nature of those interests is shown in Figure 18.1 in the form of a Venn diagram. Some

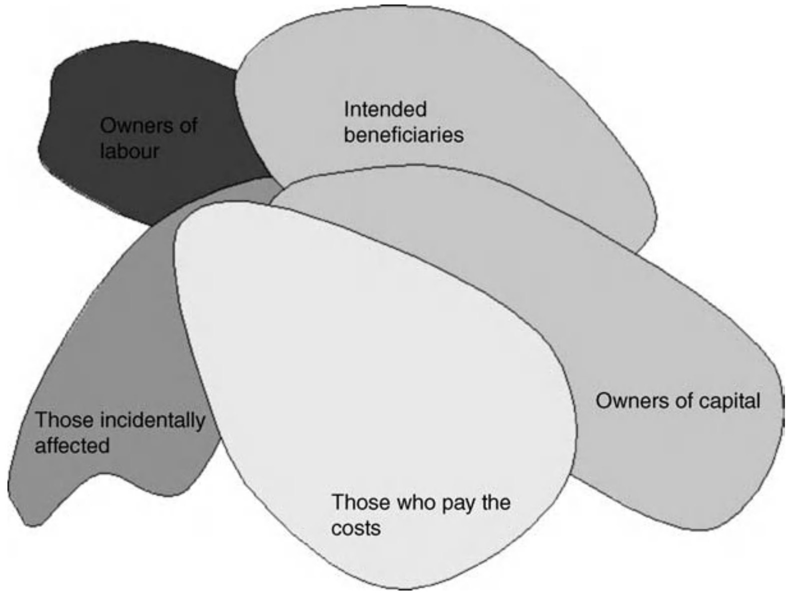

Fig. 18.1 Who has an interest?

individuals will have more than one interest (e.g. they pay towards the cost and they are a consumer of the service provided). Depending on the degree of overlap between the different sets, the conflicts of interest are internalized to each individual or organization in that intersection set or left to be resolved through collective choice.

So, for example, whilst other people in the country share a common interest in being a taxpayer, and so share in the cost of implementing a river management scheme, some will also visit the area concerned for recreational or other purposes, and some will have a concern for the environmental benefits and costs of the proposed intervention strategy. Within each of these groups, there are further subsets with differing interests; for example, the interests of those who provide equity capital for a project are not identical to those who provide loan capital. Different institutional forms generally result in quite different patterns of overlap and distribution (Green 2007). For example, whereas the costs of protecting against river and coastal flooding are shared across the UK taxpayers as a whole, the costs of protecting against sewer flooding are wholly borne by the charge payers of the relevant wastewater company. Depending upon the degree of overlap between the different sets, the conflicts of interest are again internalized to each individual or organization in that intersection set, or left to be resolved through collective choice.

In the specific case of flood risk management in England, Figure 18.2 shows the initial breakdown of interests. In addition to those at risk of flooding in a specific area, and the taxpayer who will bear the costs of any scheme, there are two other key groups: those at risk elsewhere in the same catchment, and those at risk elsewhere in the country. The first group has a key interest in that any scheme in one area may also affect the risk elsewhere in the catchment. The interests of the wider group are affected because given limited resources, a scheme undertaken in one area will mean that schemes in other parts of the country and catchment will be delayed or not implemented at all.

Those stakeholders who are engaged either represent their own personal interest or claim to represent some wider interest. Traditionally, the concept of legitimacy (Weber 1947) revolved around the nation state and was concerned with

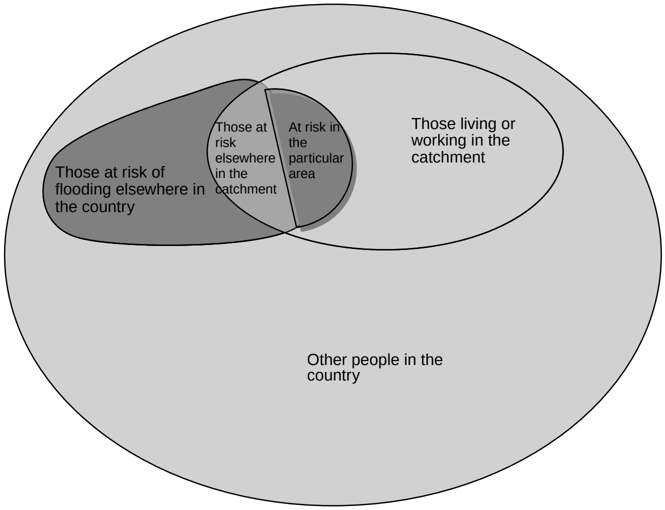

Fig. 18.2 Interests in riparian flood risk management.

whether the government acted legitimately. It is much less clear what legitimacy means in an era of organizations acting internationally (whether those organizations are companies, international agencies or NGOs) and of local stakeholder engagement.

Legitimacy may, however, be characterized in terms of broader societal agreement, or social norms, as to who should have power, what forms of power they should have, and how that power may be exercised and to what ends. Because of the importance of procedural equity in achieving the benefits of collective action, the whole of society has an interest in any single decision process lest that decision violate the social norms of procedural justice.

What might be argued is that the internal processes of any group claiming to represent a wider interest must at least replicate the procedural standards of the stakeholder engagement process. That is, each group taking part must use its power in an accountable way (Lloyd *et al*. 2007). The obvious form of power in western society is democratic election, and hence a clash of legitimacy can occur between unelected community groups or NGOs and elected bodies. This clash is in addition to wider differences between a stakeholder who is promoting a specific interest and the duty of elected bodies to resolve the conflicts between the multiple interests. 'People-orientated' NGOs, such as those concerned with a specific local community, poverty alleviation or human rights, clearly should be accountable to those whose interests they claim to represent (Lloyd 2005). More problematic are the NGOs that claim to stand for the environment but lack a clear chain of accountability beyond their members. Therefore, stakeholder engagement raises another series of difficult questions (Green 2003):

- How representative are those involved?
- What is the relationship between unelected stakeholder groups and democratically elected bodies?
- Do the processes through which the stakeholders reach a conclusion comply with the requirements of procedural equity?
- What are the obligations that follow from being included in the decision process? In particular, are the stakeholders bound by the decision of the group? What entitlements and obligations follow from a deliberate choice not to participate in the decision process?

That someone or some group can establish a moral claim to be engaged does not mean that they will or can engage. The time requirement, and the timing of the engagement process, as well as lack of resources may exclude some from participating. In particular, women have been excluded by such considerations as well as by the formal excise of exclusionary power (Gender and Water Alliance 2006).

Equally, those who stand individually to make large gains or losses are more likely to consider that these potential gains or losses exceed the costs of participating. The role of NGOs and other organizations is thus to reduce the cost of participating, indirectly, to those who are members of such bodies. But the formation of NGOs does not necessarily match the interests of everyone who has a sustainable claim to involvement. If, for example, the costs of any river management or other scheme are borne by the general taxpayers, then unless their interest is represented, the other stakeholders can enjoy spending other people's money in order to resolve their conflicts.

Why does Justice Matter?

Human society has been dominated by concerns that individuals should behave ethically and collective decisions should be based upon justice. What has been argued over the millennia has been the question of what is the appropriate ethical framework, and what constitutes justice? What function therefore does justice serve? The first apparent purpose of justice is to avoid all conflicts being settled by physical power; the second, related, purpose is to ensure the continuity of the community. As Wenzel (2001) argued: 'Justice serves to maintain the status and values of one's group. People strive towards justice, even at the cost of their personal outcomes, because justice strengthens the values of their group and thus contributes to their social identity in terms of that group'. If there are gains to be had from cooperation or collaboration, then it is necessary to establish a system where short-term gains and losses do not prevent the achievement of those long-term gains. Both humans and other species seem to have learnt the lesson that over the longer term the gains from cooperation or collaboration are such that they must be preserved (Ridley 1997). Justice provides the base for the continuity necessary for this collaborative activity.

What is Justice?

Trying to define justice has consequently been a major theme in human thought both in jurisprudence and philosophy, whilst collective decisions are often centred on debates as to what is just, fair or equitable. For practical purposes, what is important is that justice - and what constitutes justice in a particular case - is contested.

The definition of justice can be approached from three positions:

1 defining an initial state (e.g. Rawls 1971; Noznick 1974);
2 defining an end state (e.g. Bentham 1970);
3 defining a process by which to reach a decision.

The first two approaches offer the apparent possibility of a rule by which individual decisions can be reached, particularly if the initial or end state can be ascribed either to natural law (e.g. Noznick 1974) or the requirements of a deity. An alternative approach is that the state arises from a social contract – a binding, overarching agreement in forming a community. This is the approach to which Rawls is the most recent and influential exponent.

There are, therefore, differing conceptualizations of justice and so the debate in a particular situation frequently centres about which concept of justice to apply. That debate is often made more difficult insofar as a simple rule is unlikely to be seen as yielding equitable results in every single case – in English law, the law of equity developed to deal with cases where the rule of law was not seen to result in justice (Worthington 2003). Conversely, more complex rules are likely to give different results in the same instance; thus, Sen (1992) argued that the problem with equality is that the achievement of one form of equality is likely to preclude the achievement of another form of equality.

What debates about justice share are:

1 Justice is the application of some universal principle consistently across cases; so Green (2003) summarized the concept of justice as 'a moral principle consistently applied'. What is then argued is: which moral principle ought to be applied, and what constitutes consistency?

2 The use of antithesis to define the core of justice. Thus it is often easier to specify what justice does not include than to specify the nature of justice itself. Hence, a common approach to a definition of justice is by antithesis; by defining what justice excludes. For example, the oath of office of judges in the UK defines both what they will do and what they will not do: '. . ..and I will do right to all manner of people after the laws and usages of this realm, without fear or favour, affection or ill will'. Hence, an important characteristic of justice is what it excludes; thus, the statues of the Roman goddess of law, Justicia, frequently found on court buildings, are often shown with a blindfold as well as a set of scales.

What both the concepts of consistency and the wide use of antithesis to define justice share is that an important aspect to justice is to distinguish between differences that should be taken into consideration and those that should not. Lloyd (1991) asserts that a condition for justice is that: 'all those in the same category shall be treated as equal', that is, in accordance with the categories laid down by law and not in either an arbitrary or biased manner. Then a key question becomes defining the 'same category': what makes those within that category similar and different from others.

Is Justice Possible?

For the state approaches to the definition of justice to be workable, there are two conditions that must be satisfied:

1 the approach must be applicable across all possible instances; and

2 it must produce a unique answer to every possible instance.

These ambitious expectations and uniqueness has to be bought at the cost of reducing the differences that are taken into account. In addition, in a changing environment, equality is dynamic: what achieves equality at one moment may not consequently result in equality at a different time since equality is a balancing act. Consequently, equality exhibits path dependency (Green 2007): what should be done now depends upon what was done before, and therefore what should be done in a particular choice depends upon the sequence in which choices are made. The underlying assumption in the state approach is that there is nothing left to learn; only then could we expect to recognize a universally applicable, uniquely determining principle.

Equality is also about what differences ought to be taken into account and conversely what differences ought not to be taken into account. Equality asserts that some differences ought not to be taken into account (e.g. in gender, ethnicity) but allowance should be made for other differences (e.g. the formal recognition of disability).

When considering flood risk management, there is a wide variety of initial inequalities that might be taken into consideration (Johnson *et al.* 2007). Those starting inequalities include the:

- Type of flooding, e.g. groundwater flooding, sewer flooding, river flooding, flash flooding and so on.
- Likelihood of flooding.
- Severity of flooding – depth, duration, velocity and load.
- Cost of reducing the risk; the cost per property typically depends upon the local geometry so that the cost of protecting a property will vary dramatically.
- Resources (power) available to those at risk to ameliorate the risk, notably income but including the form of tenure.
- Resources available to cope with an event; they differ depending on the extent and forms of power available to them.
- Resilience of those at risk – those at risk differ in the power they have to draw on different resources so as to recover from a shock such as a flood.
- Contribution of those at risk to wider society. Traditionally, flood risk management focused on agriculture because when more than 50% of household income was spent upon food, food shortages and the resulting price rises meant that people starved.
- Power to choose where to locate. The poor get the last choice of land, and that land is often exposed to one or more hazards be these pollution, slope instability or flooding.

In turn, those starting inequalities necessarily produce a finishing inequality of some kind if only in that some households will have more money spent upon reducing the risk to them than will others. Hence Ramsbottom and Green (2004) took some apparently reasonable rules for ensuring consistency of treatment and tested the consequences of applying each across a variety of case study sites. Not surprisingly, the course of action adopted depended upon the rule applied.

Procedural Justice

In practice, substantive justice is unlikely to be possible in any single decision taken in isolation (Green 2007, 2008). Only across an array of choices is it likely to be possible to achieve substantive justice; only across such an array of choices is any single stakeholder likely to achieve a satisfactory outcome. Hence, it is crucial to chain choices together; that which makes it possible to bind them together, that which provides continuity, is procedural justice. The importance of the outcome of any single choice is therefore what it tells us about the application of procedural justice. It is procedural justice that enables each stakeholder to view the outcome in terms of the chain of choices rather than focusing on each choice in isolation.

Procedural justice is both preclusive and prescriptive. In the negative sense - the definition of what justice is *not* - procedural justice is about what powers may be used by whom for what purpose - those differences in power that will not be taken into account. Thus, those stakeholders who have more power than some other stakeholder may be precluded from using that power. The judges' oath specifies some of the powers that may not be used in a trial.

But again concepts of procedural justice can differ both between individuals and between contexts (Wendorf and Alexander no date). Characteristics of procedural justice that have been proposed (Thibaut and Walker 1975; Leventhal 1980; Tyler and Lind 1992) include:

- Bias suppression/neutrality – applied in a manner that is both unprejudiced and without self-interest.
- Accuracy – the procedures succeed in their own terms and are based upon accurate information.
- Correctability – the opportunity to appeal.
- Consistency – in application across like instances.
- Representativeness – all affected should be considered in the decision.
- Ethicality – the decision should be made according to prevailing ethical standards.
- Voice/process control – are the interested parties given a full voice?
- Standing – are the interested parties respected as people?
- Trust – legitimacy.
- Decision control – do the interested parties have any influence on the decision?

Other literature stresses the importance that the procedure protects the worth and dignity of those involved in the adjudication (Lind and Tylor 1988). In this context, the attractions of the English Common Law approach (van Caenegem 1988) become apparent:

1 the use of precedents gives consistency across cases and over time;

2 but because precedents are not absolutely binding, there is scope to both learn over time and adapt to different circumstances; and

3 the extensive appeals process tests the procedural justice of each case at several stages. In criminal law, this has developed to review past cases as well, so that past failures can be retrospectively reassessed and corrected.

Social Relationships

How we decide, what we do and how we pay for it are all statements of social relationships. Hence, in debating how to manage the risks of flooding, we are also either implicitly or explicitly arguing what should be those social relationships. Conversely, how we decide, what we do and how we pay for it are the expressions and articulations of social relationships and justice in particular.

Similarly, a technology manifests social relations. Therefore, changing what we do implies a change in the appropriate nature of social relationships. Making such a change is an expression of power.

Thus, a contested area is what ought to be the relationship between the individual and the collective, an argument that underlies the arguments as to the nature of substantive justice (Pettit 1980). There are two quite different conceptualizations in France and Germany. The preamble to the Constitution of France asserts that: ≪S12 La Nation proclame la solidarité et l'egalité de tous les Français devant les charges qui resultent des calamités nationales». This is in contrast to the recent Flood Law in Germany, which has a much greater emphasis upon the individual responsibility of those at risk: 'Within the bounds of possibility and reasonability, any person potentially affected by a flood is obliged to undertake adequate measures to prevent flood-related risks and to reduce flood damage, particularly to adjust the land use to a possible risk created for humans, the environment or material assets through floods' (Article 31 S2 Act to Improve Preventive Flood Control 2005). The French approach implies state intervention, large-scale engineering works, and compensation for the losses experienced in floods. Conversely, the new German law implies a greater emphasis on floodproofing, resilient and robust construction, land use controls and flood warnings, and also commercially based insurance. Thus, the debate between 'structural' and 'nonstructural' flood risk management strategies is at least as much about what ought to be the relevant social relationships as it has been about the effectiveness of different strategies. To the extent to which the debate is presented as purely a technical question, it is misrepresented.

One key aspect of social relationships is: who should pay for flood risk management? The German model reflects the 'beneficiary pays' principle. That approach is itself in part a reflection of the neo-liberal ideology that seeks to diminish the role of the collective and exalt the role of the individual. It specifically seeks to minimize the role of the state. But, since a significant fraction of those at risk are flooded by 'other people's runoff', this would argue for the application of a second model – the 'polluter pays' principle. However, in most countries, a third model is adopted: most or all of the costs of flood risk management are paid for by the general taxpayer. This means that the general taxpayer is an important stakeholder in the decision process. There is empirical evidence, moreover, that the public at large is prepared to pay to protect others (Shabman *et al.* 1998). The underlying question is, then, why is the public prepared to contribute rather than seeking the application of the beneficiary pays or polluter pays principles? Unfortunately, as North (1990) observed, these are not questions to which economics has as yet any theoretical or practical answers. It also means that neoclassical economics, by focusing upon how much the beneficiary is prepared to pay in order to gain the benefit, is addressing the wrong question.

The role of the 'expert' is also changed: whereas engineers and economists used to talk in terms of determining the optimum solution, this is no longer appropriate. It is now the stakeholders who have the responsibility to determine which is the most appropriate option. Moreover, it can be argued that it was always inappropriate for 'experts' to claim to determine the optimum solution, that this should always have been the responsibility of politicians since they are elected for the purpose of making societal choices. The nature of these changed roles has yet to be worked out: how science can best be used for informed decision-making (Defra 2006).

Stakeholder Engagement as a Process

Stakeholder engagement has to be done and done successfully. It is clearly a process; it is equally obviously a social process; and we have argued above that it must be a learning process through which the stakeholders seek to resolve the conflicts that make the choice necessary and become confident that one option ought to be preferred above all others. Thus, stakeholder engagement is increasingly understood in the form of 'social learning' (Craps 2003; Ison *et al.* 2004; Adank *et al.* 2006). Furthermore, stakeholder engagement has to be understood not just as a learning exercise in relation to a single decision but also as learning how to improve the process. Any stakeholder engagement process involves learning about the process and so it should be reflective.

To be successful, it has to be understood as a social learning process largely undertaken through language use rather than as a mechanical exercise involving the communication of information. Since it is a process largely conducted through language, it is also necessary to recognize the nature of language and that language use is itself a social act (Berger and Luckman 1967). The stakeholders will be conducting conversations with each other; it is necessary to avoid regarding language use as no more than a form of communication or that the content of that communication is information. The purpose of conversation is to influence either the self or the other.

It is also necessary to understand the nature of language itself. Attempts are sometimes made to provide definitive definitions of such terms as 'risk' or 'vulnerability'. Indeed, early philosophers of language (Wittgenstein 1961) and semioticians (de Saussure 1983) argued that there should be a one-to-one correspondence between a word and its meaning. This gave an unwarranted appearance of neutrality to language when a major purpose of language use is to influence others (Hajer 1995). Secondly, it is the permeability of language and the ability to use it in the form of metaphors and analogies that make language useful (Wittgenstein 1958). Hence, any attempt to formulate definitive definitions of words is to misunderstand the nature of language.

As a social process, it is necessary to build both the community of stakeholders and the social process (Figueroa *et al.* 2002). Classically, communities are described as a group in 'communion' with each other so conversation defines a community. Many of the exercises proposed for stakeholder engagement, such as those proposed by Chambers (2002), are centred on community building, and there is generally a strong emphasis on collective action to build a community. Such approaches include building models of an area, or drawing a map, or putting up stickers on a wall on key issues. Successful stakeholder engagement techniques seek to promote the creation of group cohesion; as Aristotle (1955) argued, virtue is a matter of habit. In turn, the decision-making approach will tend to seek progressively to eliminate options rather than to find an 'optimum' solution, and to focus initially upon areas of agreement rather than on issues about which there is a conflict.

The engagement process will become governed by a system of implicit or explicit rules, and those rules can strongly affect the success of the process. Thus, in jury trials, starting deliberations by setting out to agree what are the key evidential points upon which establishing guilt depends has been found to be more 'successful' than going around the table asking for initial votes on guilt (Darbyshire *et al.* 2002).

The frequent emphasis on 'leadership' as promoting successful stakeholder engagement shows that the importance lies in skills rather than the adoption of techniques. Those skills are those of interpersonal relations, such as in conversation (Stone *et al.* 1999), emotional intelligence (Goleman 1996), intercultural communication (Lustig and Koester 1993), and team-working (Handy 1999).

What is Successful Stakeholder Engagement?

The purpose of stakeholder engagement is to do it successfully. Hence, a starting question is: what do we mean by success? What the appropriate criteria are for assessing project appraisal techniques has a history (Penning-Rowsell *et al.* 1992), but in this context some of those criteria refer to making and implementing 'better' choices where what may be meant by 'better' has already been discussed.

We measure success by change: change from what otherwise would have occurred or that occurs elsewhere. Hence, measuring success is centred on measuring some form of change, and the question is: what are the factors in which we hope to observe change? Since we want to do 'better', ideally we want to assess the differences in outcomes over the long term. But a problem is that we have to make this evaluation of success in the short term in order either to take corrective action if the particular instance is deemed a failure, or, if it is a success, to apply those lessons elsewhere. This means that it has to be possible to make these evaluations early on and on the basis of quite preliminary assessments when the outcome may be long term. Thus, outcome measures have inherent limitations (Johnson *et al*. 2010). Nevertheless, work by Beierle (Beierle and Konisky 2001; Beierle 2002) shows positive gains.

So, the second way of measuring success is in terms of process. Here, continuity has a claim to be the most important criterion. If, as argued earlier, the only means to deliver substantive justice is by chaining choices together so that substantive justice is delivered as a whole, then piecemeal approaches will often fail. So, any process of making choices that makes one choice and is then abandoned is a failure. Therefore, its success in any one choice is measured by the desire of the participants to repeat and replicate the process for future choices. This will occur to the degree to which each of the participants felt the process had been a success. Process success is thus important to the extent to which it promotes continuity.

Defining the key criteria as change and the sustainability of the process, then we should be seeking to measure change amongst the stakeholders' engagement: the extent and direction of the changes that occurred during the process. Thus, it is appropriate to track over the time, starting before the engagement process begins (Tunstall and Green 2003):

1 The stakeholders' assessment of their own knowledge and skills in relation to the choice.
2 Their assessment of the knowledge and skills of the other stakeholders.
3 Their assessment of what are the critical issues involved in the choice.
4 Their assessment of the attitude of the other stakeholders towards the process.
5 Their assessment of the contribution of the other stakeholders to the process.
6 Their assessment of what the other stakeholders want out of the process.
7 Their attitudes towards each of the other stakeholders.
8 Their personal or organizational preference as to the nature of the course of action that should be adopted.

Such indicators can usefully be included on a regular basis within the engagement process itself. If the attitudes towards one stakeholder of all the other stakeholders become increasingly hostile over the course of engagement process then either that stakeholder is making very effective use of power to force the other stakeholders to adopt the option preferred by the stakeholder in question, or that stakeholder is being very unsuccessful in changing others in a way that supports their case. If it is the latter, then the knowledge of their failure would offer them a chance to change tactics.

In addition, it is appropriate to seek the stakeholders' evaluation of the process itself (Green 2003):

1 Was the process fair and equitable?
2 Was each stakeholder treated in a fair and equitable way? Were their views given due consideration? Were their contributions valued?
3 Was a sufficient range of options considered?
4 Did any individual stakeholder or group of stakeholders impose their views upon the group as a whole?
5 Was adequate technical support and information made available?
6 Did the process result in any change or was the decision effectively already made?
7 Did you learn anything from the process?
8 Did you derive any personal satisfaction or benefits from the process? Would you be prepared to repeat the experience?

Conclusions

Stakeholder engagement is about mobilising social relationships for collective choice. It is thus simultaneously about what those relationships are and what they should be, and the pursuit of effective practice in them in order to deliver sustainable development. However, the shift to stakeholder engagement simply makes what was hidden open; choices always involve conflict, power and the restriction of power by justice. In turn, choices are frequently difficult and in some cases we should expect there to be neither an agreed nor an obviously preferable option. That we have to do this in the face of change in order to make change, and hence in the face of uncertainty, makes the task more difficult. Thus, we have to become better at social relationships whilst not expecting that the result will always be satisfactory whilst simultaneously learning how to do better.

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19 Flood Risk Communication

HAZEL FAULKNER, SIMON McCARTHY
AND SYLVIA TUNSTALL

Introduction

The changing role of risk communication in flood risk management

It is has always been widely recognized that effective communication about flood risk is integral to flood risk management (D. Parker, personal communication). Recently, however, flood risk management (FRM) in many western countries has gone through considerable change, expanding beyond a focus on flood defence to a period in which a wider range of professionals and their communities will become more engaged with floodwaters during flood events than had previously been the case (Hall 2003). Whilst these policy changes take time to become reality (and in fact in practice most flood risk in the UK is still addressed by the traditional flood defence methods), nevertheless there is now a noticeable shift of emphasis away from asset management, towards an approach that requires 'Making Space for Water' within residential landscapes (Department for the Environment, Food and Rural Affairs 2004). This means that risk is increasingly addressed through spatial planning and the control of development, through 'building in' resistance and resilience in property in flood risk areas, as well as by encouraging enhanced resilience within communities.

As Parts 2 to 4 in this book testify, flooding is a very varied phenomenon. For instance, there are significant differences of causality and intensity between coastal and fluvial flooding. Floods with differing causalities have extremely varied event timings, particularly extreme variations in event onset, consequently requiring differing communication strategies. High tides around which flood risk peaks at the coast can be planned for, and at least partially anticipated in advance. In the river context, however, variations in catchment size above the riverine setting affects communication possibilities. Small upstream catchments, where generated discharges may be proportionately smaller,[1] experience rapid-onset floods where warnings may not be possible. In comparison, on floodplains in the middle and lower reaches of large river catchments, with good radar warnings, onset can be anticipated several hours in advance. In urban areas, rapid-onset chaotically generated pluvial flooding allows little in the way of anticipatory warning time. Because all flood risk professionals are now required to address all forms of flooding, including coastal, surface water, groundwater and pluvially driven sewer flooding, their new roles now demand a much wider range of risk communication activities beyond the simple flood warning.

In general, flooding is a high-intensity, low-frequency phenomenon, so that at different stages in the 'hazard cycle'[2] (see Fig. 20.1) and over different timescales, communication activity between professional groups and the flood-affected public takes place in different ways. Between flood events, floodplain mapping and preparedness

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[1] In England and Wales there are reported to be around 450 rapid response rivers.
[2] Referred in Chapter 20 as the 'disaster management cycle'.

dominate. Across Europe, in response to the recent EU Floods Directive, a wide range of professionals are adopting flood risk maps as the main risk communication tool for the proactive management of the flood-affected public (see Fig. 19.1). For emergency response professionals, for instance, some activities overlap in time, so the raising and issuing of flood warnings to floodplain occupants in the immediate period preceding an event in real time has to be maintained alongside preparedness-raising and capacity-building procedures.

In the UK, existing medium-term climate change models predict increases to flood size, frequency and intensity (Baxter *et al.* 2001). In this context of change, flood risk professionals of all types working in any particular geographical location will see a shift in the agendas as they struggle to interface with a much broader range of hazards. They will be expected to communicate with a public audience that will increase in geographical scope to include many of those currently designated as occupying low-risk areas (living behind existing defences or outside the currently defined 1:100 flood-risk zone). So flood risk professionals need to communicate with a much broader group of other professionals with different but parallel agendas ('intra-professional communications').

Communication between whom?

Traditionally, risk communication models concentrate largely on communications between professionals and public (e.g. O'Neill 2004; Wardekker 2004). However, because of the policy shift outlined above, the flood risk communication realm is now populated by different groups of professionals who need to communicate effectively with one another in order to articulate flood risk

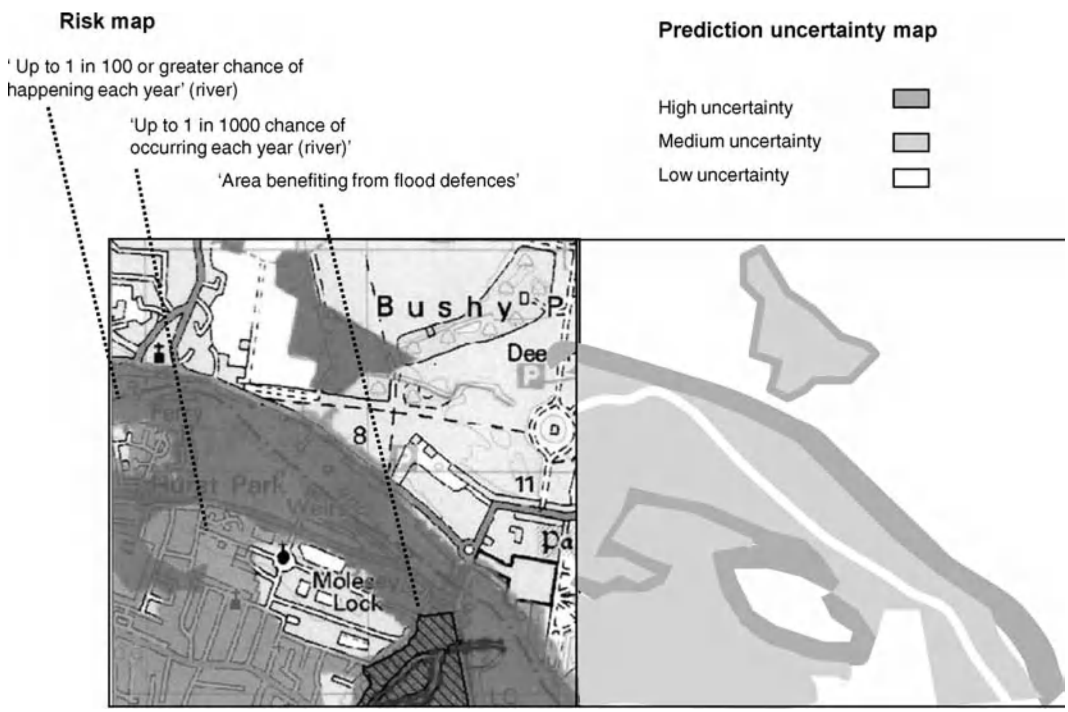

Fig. 19.1 Left: Indicative floodplain maps, as available on the UK's Environment Agency (EA) website. Right: Pattern of uncertainty associated with the risk estimation. (See the colour version of this figure in Colour Plate section.)

in an unconfusing manner to both the public and each other. Consequently we have chosen to broaden the concept of flood risk communication here, to include the information flow between all involved. We identify three categories or groups for whom communication challenges differ (the three categories can be seen as the shaded boxes in Fig. 19.2).

Firstly, those at the **source** of information about flood science, whom we refer to here as 'scientists', being those whose work involves developing the science base and new knowledge that underpins flood risk management. This group includes meteorologists, hydrologists, flood modellers, and weather and flood forecasters working at the UK's Met Office or within the Environment Agency (EA) itself, as well as economic modellers, and hydraulic engineers developing asset failure models. Secondly, there is a group of non-science professionals whose work mainly involves delivering flood risk management options for society. From the perspective of the movement of scientific knowledge, these could be argued to have a **translatory** role (Faulkner *et al.* 2007). We refer to this group, which includes insurance agents, planners, managers within the utilities, as well as the professionals in the EA, as 'flood risk professionals'. Thirdly, we include the 'public', the **receptors** of flood risk information. The term 'public' is not confined to occupants of known flood risk areas, but also covers other non-professional stakeholders such as community groups and flood action groups. In fact there is no longer just one 'public' but many different audiences. These include all those with an interest in flood risk management and communication who are

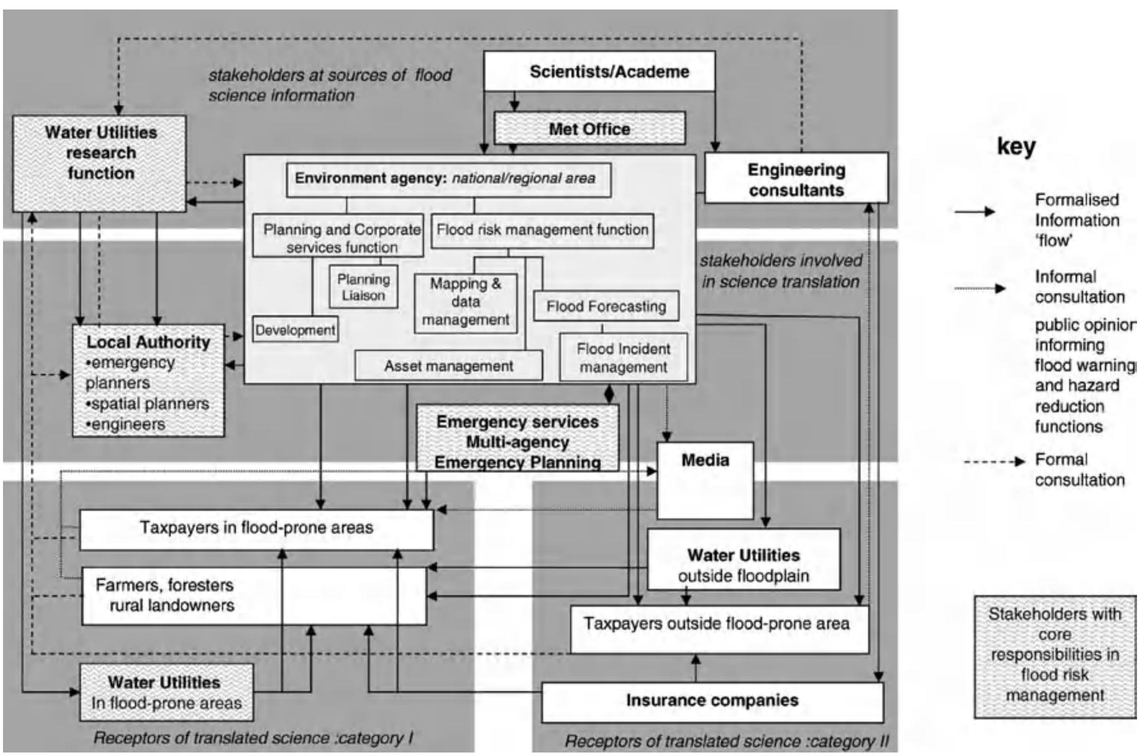

Fig. 19.2 Pattern of information flow in flood risk management in the UK. The various flood risk professionals involved are suggested to have an uncomfortable intermediary translatory function between the scientists at the source of flood risk science and the lay audience to whom they are formally or informally obliged to communicate about flood risk.

not defined as scientists or professionals, and many of them well-informed and increasingly engaged. Additionally, these categories sometimes overlap, as the EA in England and Wales employs professionals whom we would think of as scientists but whose role also embraces risk translation; moreover, the professional journalist is hard to categorize as the media have a parallel and largely reflexive relationship with both the public and flood risk professionals.

In this chapter we also choose to include consideration of the internal communications that occur between these flood risk professionals themselves, referred to here as 'intra-professional' communications. In England and Wales, the responsibility for flood risk management falls largely within the professionals of the EA. But because of the change in policy outlined above, EA managers are now required to communicate with (and be open to) influence by the views of not only the public in flood-affected areas, but also with this much wider parallel professional stakeholder community (Defra/EA/2002, 2004). These other professional stakeholders include emergency service managers, for whom emergency training and exercises remain a high-level professional activity. They also include spatial planners, the insurance industry, managers of utility companies, and journalists, who clearly regard themselves as risk communication professionals. At the same time, the field of flood risk communication has witnessed an increase in the number of local flood action groups, as well as the formation of a National Flood Forum, which are now also involved in the process of flood risk communication.

All these professionals have responsibilities not only to communicate effectively at the public interface, but also to act as a source of information to each other and other professional agencies, as suggested by the pattern of linkages in Figure 19.2. All of this requires new communication skills channels, and new tools fit for the internet age (Table 19.1).

The rest of this chapter covers three topics. Firstly we explore the challenges involved in the incorporation of scientists' formulations of floods (rainfall and runoff models) into the work of flood risk professionals. Secondly, the 'intra-professional' communication challenges that exist between flood risk professionals are briefly explored. The final section considers professional communications with the public. The chapter draws on research on flood risk communications in the UK carried out as part of our work for Research Priority Area 7 for the Flood Risk Management Research Consortium.

Communications between Science and Flood Risk Professionals

In all the professional settings identified above, the incorporation of scientific formulations of flooding has expanded enormously, and theoretically these advances should make professional exchange easier and faster, as this knowledge is

Table 19.1 Communication tools for risk and/or uncertainty

Normal science tools	Post-normal science tools	Social tools
Rainfall-runoff models plus GLUE methodology (Generalised Likelihood Uncertainty Estimation methodology)	Softer communication tools, e.g. NUSAP tools (Copernicus Institute)	Focus groups
Radar prediction/forecasts and `nowcasts'	Warnings with uncertainty weightings	Open consultation vehicles
Event-frequency curves	Risk zonation maps weighted for error/uncertainty	Newspapers,TV and radio
Probability density functions		Public hearings
Bayesian tools		Leaflets
		Booklets
		User-friendly webpages with nested interrogation levels (progessive detail)

transferred and incorporated into both planning and warning procedures. Previously, the incorporation of models into flood mapping concentrated on fluvial (upstream-downstream watershed-based) models and to some extent coastal floods. Now, partly in response to the Pitt Review (Pitt 2008), scientific modellers are extending their activity to include surface water runoff and urban drainage issues, such as contaminated sewer surcharges. Professional agencies are continuously and reflexively incorporating the science of FRM, and then tailoring the scientific end of flood risk models as communication strategies and tools.

Yet given the rapid pace of these changes, the models are sometimes described as having developed beyond the capacity of the professionals to incorporate them, and this applies particularly to the incorporation of uncertainty tools (Kinzig *et al.* 2003). Certainly, there is currently somewhat poor connectivity between radar warnings, even top-of-the-range radar-based prediction models, and their ability to improve communications to the public during real events. Real-time radar predictions of flood onset are also poorly interfaced with the models that predict inundation patterns, depths and velocities, although as this book testifies, these models do separately exist. Enhanced co-operative working between the Met Office and the EA will improve communications (Purdey and Davies 2008). Faulkner *et al.* (2007) argued that translating complex and uncertain science is a considerable additional challenge for FRM professionals.

In Figure 19.2, the flow of scientific information, in the form of meteorological, flood and inundation models couched largely in mathematicized language, is portrayed as suffering what Mansilla *et al.* (2006) would term a 'serious translational gradient' by the time this information interfaces with the public. Since scientists, professionals and the public use the terms 'risk' and 'uncertainty' and 'probability' in entirely differing ways (Holmes 2004), the problem is partly semiotic (Morgan *et al.* 2002; Holmes 2004; Leiss 2004; Pappenburger *et al.* 2005; Wilby et al. 2008). The EA (Environment Agency 2004b) is now moving forward on several fronts to create opportunities for a fresh look at the strategies and tools that they (and other agencies) use, both in the context of real-time emergency response to floods and in flood risk mapping conducted by planners and engineers.

Science and risk communication tools and methods

Some of the tools available at the science–professional interface are listed in the first two columns of Table 19.1. Of these, floodplain maps are the most obviously useful risk communication tool that flood risk managers can use for forward planning. Their development is underpinned by a widely applied classical scientific formulation of risk. This formulation calculates risk at any time, t, as the product of the probability of an event (based upon the statistical properties of the long-term event series), and the consequences of that event (Kron 2002). This is most often expressed in the form:

$$R_t = H_t \times V_t$$

where R_t is the risk estimate at time t, H_t is hazard or probability, and V_t is vulnerability or consequence (which, for floods, is often expressed in economic terms as a cost of damage). The generic term 'H_t' is taken here to embrace distributed physical models of rainfall events, or storm surges, or floodplain inundations for event at time t. The scientists included in the papers presented in this book are primarily concerned with modelling a range of hydrological and/or meteorological processes through time t (ΣH_t) at various recurrence probabilities. Models can simulate a single runoff or rainfall event and make real-time predictions. In the domain of flood risk management, however, most physical scientists were taught about flood prediction and management as being underpinned by probabilistic extreme event theories. In these, the hazard H_t is then defined in terms of a discharge frequency curve (Pappenburger *et al.* 2005) that can be derived from existing long-term records (at least where such records exist).

Flood science underpinning planning and in warnings

Traditionally, then, H_t is the central component of the traditional formulation of risk, R_t, and in floodplain settings above is described by a discharge frequency curve derived from a long-term hydrological record. The consequences part can be assessed for individual hazard events by using the peak discharges as modelled with the aforementioned physical models, to give assessments of floodplain inundation for that event, which can then be costed for damage using depth–damage curves to give an economic estimate of the consequences of such an event (Penning-Rowsell *et al*. 2005). Integration over the cumulative distribution of events provides an estimate of total risk, ΣR_t, at a particular location. Whilst there is not sufficient scope here to review the vulnerability literature, increasingly other measures of vulnerability can also be used, for example measures of societal vulnerability. Thus total risk, $\Sigma R_{t,}$ is sensitive to both spatial and temporal variations in both H and V. Total risk calculated this way can be mapped, and used to define zonation bands for use in prioritization and decision-making, especially in a floodplain context (e.g. the high, medium and low flood risk zones used by professional planners). Additionally, the static 1:100 maps etc. can be replaced in the risk assessment by real-time one-dimensional (1D)/two-dimensional (2D) simulations to assist decision-making (as discussed in Chapters 12 and 13 of this book).

This classical formulation of risk as the product of the probability of an event and the consequences of that event is still widely used, despite arguments that the stationarity of the temporal series (and therefore its potential to be extrapolated into the future) has to be questioned in the context of climate change predictions. However, in the sense that both H and V are estimates, there are considerable uncertainties embedded in the EA flood risk maps available for the 1:100 and 1:1000 events – it is an inconvenient truth that flood hazard maps are as certain as the models used to generate them (see debate in Lavis *et al*. 2003). One solution is to incorporate local knowledge more fully, to enhance and validate the maps.

Pappenburger *et al*. (2005) explored the wider range of tools that is available for the articulation of both future flood risk and uncertainty by hydrologists, oceanographers and meteorologists in real-time situations. Their optimization for differing professional challenges in FRM was explored by Faulkner *et al*. (2007). In the more constrained context of a real event, however, FRM professionals rely much more on the radar/rainfall ensemble interface, and it is here that the role of uncertainty tools has seen the most progress.[3] These tools and their application are discussed in much more detail in previous chapters (see Chapters 7–9 and 14).

Intra-Professional Flood Risk Communication

Quite apart from the translation of scientific formulations of risk as a means for communication tool development, professionals frequently communicate between themselves. As part of the activities in the first phase of the Flood Risk Management Resesearch Consortium (FRMRC), the Flood Hazard Research Centre at Middlesex University (FHRC) undertook case-study research on the professional risk communications occurring in two professional settings: spatial planning and development control, and in the warnings field in the UK (McCarthy *et al*. 2008).

We looked specifically at **flood maps** and **warnings** as flood risk communication tools. The research explored the constraining effects on their incorporation into risk communication strategies of (i) the differing professional settings, and (ii) differing flooding contexts represented by three case study areas. The differing flood types we studied were described as 'simple' (slow onset, predictable floodplain flooding such as occurs in the mid-Trent basin in the UK); 'complex'

[3] Because these tools are extensively outlined in other parts of this book, readers are referred to other chapters for the details of these approaches.

(meaning of complex causality, as in urban flood contexts like the lower Thames in the UK); and 'pluvial' (adventitious rapid-onset urban floods such as occurs in Glasgow's urban area).[4] The research set out to address the following questions:

• What are the constraints on professional agendas and policies, and how do these constraining factors influence attitudes towards communication of risk and uncertainty? (variation by professional agenda).
• In differing flood-type contexts, what are the needs, strategies and communication tools presently used by professional agencies, and their strategy towards (in particular) communicating with stakeholders? (variation by flood onset type).
• How does the 'emergency cycle' affect risk and uncertainty communication?
• How can we best align views, strategies and communication tools?

At the time of the research undertaken by FHRC, great changes were taking place in both warning and planning. It emerges that given the recent development of an overarching coherent national policy for flood risk communication in relation to spatial planning, and the EA's drive for national consistency in policy and practice in all their activities, perhaps not surprisingly few regional variations were found in the professional constraints and contexts facing these groups (McCarthy 2007). McCarthy's research, designed to identify preferred risk communication tools, found that the EA's continually updated indicative floodplain maps still represent the core tools used by planners. However, the routine incorporation of new, more sophisticated modelling is often constrained by resources. The current challenge is to bring some of the new animated visual modelling tools into the planning application process itself (Tunstall *et al.* 2009). Furthermore, there were relatively few variations in the structures, strategies and communication 'tools' used in the three differing geographical ('flood onset') contexts, which we described as simple, complex (contested) and pluvial flood risk settings.

It appeared that constraints on communication in planning development were greater in the Thames area's complex (contested) flood setting than in the other geographical contexts, which may have been due to greater development challenges and local resource pressures in the locality. Other than this, the few specialized local variations reported by the professionals interviewed appeared to be dependent upon historical mechanisms and initiatives that were in place before nationally consistent approaches and mechanisms were introduced. It is important to note, however, that even with imposed national consistency, such flexibility at a local level remains an important aspect of effective communication.

McCarthy *et al.* (2007) and McCarthy (2007) considered the ways in which communication of risk and uncertainty might be optimized by interviewing senior professionals in both flood risk science and in the professional agencies identified in the central box in Figure 19.2. Communication between professionals was found to occur across all the flood management options. Findings indicated that choice of communication methods varies not just with professional context and agendas, but with both the temporal and spatial scale of application. The issue of temporal variability in communication choices was mentioned in many discussions with professionals. In Chapter 20 (see Fig. 20.1) the 'hazard cycle' is portrayed as divided into two temporal phases: before (prevention and preparedness) and after the event (response and mitigation). For flood warners, during the differing temporal phases of the 'hazard cycle', a range of different tools are utilized in warning both at the preparatory and event management phases. Technology was viewed by respondents as a key driver of progress in both flood warning and planning development. Common to both the professional planners and warners who were interviewed,

[4] In Glasgow, the institutional and pluvial flooding context and stage of development of services and approaches proved to be so different to those in the two English case studies that the focus of the research there had to be rather different. On analysis it was discovered that the focus of the Glasgow case study is directed at the development and implementation of a single important mechanism, the Glasgow Surface Water Management Plan, and so the Glasgow case study will be reported outside this chapter.

spatial probability modelling combined with decision matrices were mentioned; these have the advantage of allowing possible consequences to inform a risk-based decision support approach. Choices about communication tools and mechanisms that these professional groups employed varied more in the warnings field than in the planning field. This is obviously because of timescale variability: the aspirations of the latter group are to mitigate the consequences of an event well in advance of that event. The professional agenda of flood warners, by contrast, is largely focused around communications in the time-pressed immediacy of an event, so that their activities follow the 'hazard cycle' far more closely. In both contexts, the senior professionals in the survey saw scope for enhancing the understanding and use of scientific formulations of risk and uncertainty methods beyond the scientific community.

Communications between Flood Risk Professionals and the Public

The new emphasis on a broader definition of risk discussed in the introduction is the result of a direct challenge to a 'top-down' science-to-practitioner communication model by the so-called 'post-normal' social scientists working within the field of risk communication since the late 1980s (Gurabardhi *et al.* 2004). Early papers by Beck (1992); Slovic (1993); Kasperson *et al.* (1988); Renn and Levine (1991) and Fischhoff (1995) argued strongly that scientifically based communications that misunderstand or downgrade the importance of the knowledge, attitudes, experiences, values and perceptions of the 'risk receivers' were misplaced and inappropriate. Given the parallel shift to 'Making Space for Water' – itself a challenge to bring public and professionals much closer together 'on risky ground' (Holmes 2004) – a fresh look at communication challenges at the public interface is now underway in the UK water management industry. In other words, the arguments in post-normal science are finding new support and articulation at the public interface.

Based on extensive empirical analyses in Germany, Italy, England and Wales between 2004 and 2007, evidence indicates that there is a considerable gap between the scientific understanding of flood risk and its management on the one hand, and the risk constructions of the people in flood-prone areas, which influence their actions and behaviours, on the other (Steinfuhrer *et al.* 2007). Flood risk communication takes place in a social context (Quarantelli 1990; Mileti and Fitzpatrick 1992; Drabek 2000; Patt and Schrag 2003), such that professional sources are only one of the means by which members of the public may engage with risk prior to flooding or when flooding occurs. Risk is communicated in communities through networks of social processes, among neighbours and friends, in community and flood action groups independently of communications from scientists and flood risk management professionals.

Risk communication tools at the public interface

Table 19.2 lists the wide variety of possible tools or strategies used in flood risk communication with the public in the UK across the range of flood management options available. This elaborates on the short list described as 'social tools' in Table 19.1.

Although the importance of engaging with local people in managing flood risk is now widely recognized, it is clear from Table 19.2 that in practice most of the tools currently employed in the UK offer a limited level of engagement (either as information provision, or consultation with feedback opportunities) rather than active involvement in decision-making. Local authorities in England and Wales are required to involve local communities in their development planning processes, in which flood risk may be an issue, and this is reflected in their use of engagement tools such as workshops, and other specialized techniques. Communities and households are encouraged to develop their own community and household flood plans. However, in England and Wales, community and flood action groups are not

Table 19.2 Possible ways of communicating risk to the public based on tools used in the UK (adapted from Tapsell *et al*. Communications Audit 2006)

Flood risk management activities/options	Spatial planning		Warning and emergency response		Insurance	Flood asset management
Risk communication	**Development plans**	**Planning applications**	**Awareness raising, preparedness and emergency planning**	**Flood warning and response**		**CFMPs, strategies, schemes**
Time period (relationship to hazard cycle)	Long timescale– year (unrelated to hazard cycle)	Over weeks or months; year round (unrelated to hazard cycle)	Year round or focused in campaign periods	Over a short time period: days or hours prior to, during and after an event	Yearly renewal; claim activity in the aftermath and the recovery period	Over a period of years; may be accelerated or initiated by flood event
Flood maps and modelling	*EA fluvial and coastal flood maps. LAs' area Strategic Flood Risk Assessments may involve additional modelling	*EA flood maps. Developers' site-specific Flood Risk Assessments may involve site-specific modelling	*EA fluvial and coastal flood maps available on the internet	–	*Some insurers have developed their own modelling maps	*Flood maps and modelling:key element in CFMPs, strategies and schemes
Leaflets and brochures Newsletters, with response slips, questionnaires	*	*Developers may produce leaflets for major sites	*EA/LA leaflets on flood warning system, and what to do before, during and after a flood, flood protection	–	–	*Leaflets/newsletters describing strategy/scheme proposals
Documents, reports without/with consultation response opportunities	*	*For major developments	*Emergency plans; local risk registers on LA websites	–	–	*CFMP, Strategy, Scheme documents
Letters, notices	*Inviting comments from stakeholders	*Letters, official notification requirement for neighbours	*Letters on opting out/in to telephone warning system	*Letters on groundwater flooding	*Correspondence on renewal, premiums and claims after flooding	*Letters to directly affected residents/ businesses
Unstaffed/staffed exhibitions/ displays	*	*By developers for major developments	*Displays in libraries, supermarkets, shows, etc.	–	–	*Exhibitions in libraries, supermarkets, etc.

National media advertising, including posters	–	–	*National awareness campaigns	–	–	–
Local media advertising including posters	–	–	*Local awareness campaigns	–	–	*
National media news coverage, features, discussions	–	*Rare coverage for major developments, e.g. Thames Gateway development	*Major flood events, response and recovery; BBC Weather bulletins show flood warnings	BBC Weather bulletins show flood warnings in force and Floodline telephone number to call	*Insurers' organization issues statements on insurance policy and flood risk, e.g. on Thames Gateway development	–
Local media news coverage, discussions, features	*	*Controversial over floodplain development	*Local media coverage of local flooding raises awareness	*Local radio coverage of warnings	*Local media stories on insurance in local flood risk areas and events	*
Websites	*Development plan documents and responses on LA website. Flood maps on EA website	*Planning applications on the LA website. Community groups can monitor applications	*EA and LA website advice on warning system and what to do before, during and after flooding	EA and some LA websites provide data on flood warnings in force	*EA flood maps provide information on sites of low, medium and high risk as defined by the insurance industry	–
Telephone line: with recorded messages, e.g. Floodline	–	–	–	*EA's telephone hotline, Floodline, provides information on flood warnings in force	–	–
Staffed telephone lines	*	*Planning staff available to discuss planning applications	*	LA and EA staffed telephone lines during a flood event	*Insurance companies' staffed call centres respond to customers	*Major schemes may provide a point of telephone contact
Surveys, interviews, questionnaires	*May be used as part of development plan consultation	–	*May be used as part of awareness raising and monitoring activities	–	–	–
Public meetings/video presentations	*May be part of development plan consultation	*Possible for major developments	–	*With flood affected in the aftermath of flooding	–	*As part of CFMP, strategy or scheme promotion
Site visits with professionals on hand	–	–	–	–	–	*As part of engagement process

(continued)

Table 19.2 *(Continued)*

Flood risk management activities/options	Spatial planning		Warning and emergency response		Insurance	Flood asset management
Risk communication	**Development plans**	**Planning applications**	**Awareness raising, preparedness and emergency planning**	**Flood warning and response**		**CFMPs, strategies, schemes**
Workshops and other special engagement techniques	*May be part of development plan consultation	–	–	–	–	*May be part of engagement process
Forums, community, flood action, steering groups	*May offer discussion of flood risk issues in development plan	*May provide forum for discussion of warning system	*May provide forum for discussion of flood risk with professionals	–	*May provide a forum for discussion of insurance issues	*May provide a forum for discussion with professionals
One-to-one meetings	–	*With householders/ businesses making planning applications	–	–	*Meetings with insurers and loss adjusters	*Meetings with those directly affected by schemes
Additional official flood warning methods	–	–	*Flood wardens	–	–	–
	–	–	*Personal telephone calls/contact with EA, LA, emergency services	–	–	–
	–	–	*Sirens, loudspeaker messages	–	–	–

*Denotes employed in this option.
EA, Environment Agency; LA, Local Authority; CFMP, Catchment Flood Management Plan.

part of the long-term processes of planning for and developing emergency plans, multi-agency plans, local risk registers, and emergency training and exercises. These remain a high-level professional preserve although their outputs usually are generally available to the public on the internet. In flood warning, the use of local people as flood wardens provides a direct link between professionals and the community, which can serve to foster effective dialogue and raise awareness not only in time of flood but also at other stages in the hazard cycle. Other areas of flood risk management in which stakeholders and local people may play a more active part are Catchment Flood Risk Management Plans (CFMPs), strategies and particular local schemes as illustrated by projects in the UK and elsewhere in Europe to 'make space for water', and examined as part of the Floodscape Project (Tapsell *et al*. 2005; Sorensen *et al*. 2006).

The data in the table highlight the reliance placed on technological means for communications between professionals and the public in flood warnings and in relation to other flood risk management options. This has dangers because use of technologies such as the internet, although growing, is not universal, and the use of some relatively new and potential warning technologies, such as mobile telephones, SMS text messaging and the new range of wireless communication technologies, such as Bluetooth, may not be accessible or acceptable to some of the arguably most vulnerable members of society – the elderly, disabled and the poor (Tapsell *et al*. 2004).

In a flood event, information technologies can have a significant role in informing the public. The EA reported that over the period of the summer floods of 2007, there were 4 million visitors to its website and 206,000 calls to its recorded message service, indicating how these tools serve not only to inform the flood-affected communities, but also to raise awareness of flood risk in the wider population (Woolhouse 2008). But communications technologies are themselves vulnerable during flood events, in which electrical supply and telephone landlines services can be lost. In these circumstances, mobile telephones, battery-operated radios and computers can provide crucial, if temporary, communication mechanisms between professionals and the public. The floods in Carlisle in 2005 offer an example of breakdown of communications technologies during a flood event through loss of power supplies, and highlight the problems that may arise when too much reliance is placed on communications technologies for event management and response (Government Office for the North West 2005).

As described above, in England and Wales, EA flood maps (indicative floodplain maps) published on the internet are seen as a key tool for communicating flood risk to the public between events. However, it has already been noted that these flood maps show only fluvial and coastal flood risks. From events that occurred in Hull and Tewkesbury in 2007, it became clear that rapid-onset pluvial and other forms of flooding can occur not only within these mapped zones but also, significantly, outside them. Initial mapping of pluvially generated surface water flooding risks in England has only just been undertaken in England and Wales, and is not yet available on the internet. In addition to the national internet maps, special site-specific mapping and modelling in relation to flood management options is undertaken and may be used in communications with local people, a change that supports a shift by the EA from wider (hydrologically based) to smaller (community-based) flood warning zones that are more meaningful and relevant to local people and that hopefully will provide more targeted and accurate warnings.

Despite this impressive range of communication vehicles that may be deployed by professionals, there are many other ways in which members of the public may engage with flood risk prior to flooding and may receive a warning when flooding occurs. For example, it is well documented in EA research that people seek information and confirmation of flood warnings from neighbours, friends and relatives when flooding threatens, and they themselves warn others. In floods in Carlisle and the Northwest of England in January 2005, as many people received a warning from informal sources as from the EA's formal telephone messaging service (Environment

Agency 2005b). Informal warning systems may complement or in some instances contradict warnings from professional sources, and both formal and informal warnings have advantages and disadvantages as warning mechanisms, as Parker and Handmer (1998) have noted.

Case study: professional-to-public flood risk communication in three UK settings

In this section we draw on further quantitative case study research with local communities undertaken as part of the FRMRC Phase 1 work to explore the efficacy of some of these flood risk communication tools and strategies. Personal interviews were carried out in winter 2006–7 in the three case study areas (described above under 'Intra-professional flood risk communication'). Interviews were undertaken with residents in the 1 in 100-year return period flood risk areas in the mid-Trent, where villages were flooded in November 2000, and in Chertsey in the Lower Thames, flooded in January 2003, and with residents located in streets in Glasgow city affected by flooding in July 2002. With 100 respondents in each of the case study areas, the data were weighted to ensure equal numbers of those with and without flood experience in each area. The interviews explored residents' engagement with (i) flood risk in relation to flood awareness, preparedness and **flood warnings**, and with (ii) planning and development control: awareness of **flood risk maps**. Relevant experience of communication tools, and residents' trust in and reliance on public agencies, were also established by the interviews.

The FRMRC case studies provide evidence of the dangers of over-reliance on the internet as a communications mechanism. Reported use of the internet by household members in the case study areas ranged from 46% in Glasgow, to 45% in Trent and 33% in Chertsey, but only very small proportions of the household users (15% in Glasgow and Trent, and 21% in Chertsey) recalled accessing the internet for listed forms of flood risk or flood warning information, or guidance on what to do before or after a flood. EA research confirms a low level of use of its website for accessing flood risk information among those at risk (Environment Agency 2005a).

The FRMRC case studies also highlighted the limitations of the internet indicative floodplain maps as a way of communicating flood risk to the public. When shown an example of these maps, only minorities of respondents (13% in Glasgow, 26% in Trent and 11% in Chertsey) recalled seeing them previously for their area. The research also illustrated the importance of social networks and community sources as mechanisms for raising awareness of flood risk. People in the FRMRC case studies first found out about the risk of flooding from their neighbours, observation and flood experience as well as from formal or official sources. EA research (Environment Agency 2004a, 2005a) also demonstrates that it was more common for those at risk to have found out that they were at risk through living close to a river or coast, through previous flood experience, or from neighbours, friends or relatives than from more formal sources.

Development plans and development control are important areas for intra-professional communications, but public exposure to communications regarding flood risk in relation to development plans and development control appears to be limited. The public may become involved in major developments in which flood risk issues are being negotiated between developers, the EA and local planners only when the developments are controversial and attract the interest of the media and community groups. A very small proportion of people will become engaged in discussions about flood risk when making their own householder planning applications. In Trent, 16% had applied for planning permission at their current property, but only 3% reported that flood risk was raised as an issue for the application. In the other locations, hardly any householders were involved in householder planning applications involving flood risk.

Research also suggested that where the location was temporally in the 'hazard cycle' was a significant factor in use of tools and strategies. In the post-flood event period, flood risk managers inevitably have to engage more actively with often hostile local people through public meetings, at which they may seek to explain the mechanisms that caused the flooding and the options available for future management. Events may stimulate the setting up of special mechanisms to enable dialogue between professionals and local communities or flood action groups, as people seek to understand why flooding occurred and what can be done about it. The Flood Risk Action Groups and other community liaison groups set up in the Lower Thames following flooding there in January 2003 are examples of this. Public engagement requires time and professionals with particular skills and training. It is not therefore surprising that it is concentrated in those flood risk management options that take place over periods of months or years generally unrelated to the 'hazard cycle'.

These UK findings have resonance with wider European research. Findings from the European Commission's FLOODsite project suggest that the awareness of flood risk is very uneven across EU countries, as is the adoption of community measures to reduce exposure and risk (Steinfuhrer *et al*. 2007). Many of those interviewed in Europe do not expect future flooding to be worse than past floods (Steinfuhrer and Kuhlicke 2007; Burningham *et al*. 2008). Moreover, flood risk awareness is largely related to fluvial or coastal flooding and not appreciated in relation to pluvial or groundwater floods.

The focus of the following section is upon flood awareness and preparedness and upon flood warnings.

Public perception, and engagement with flood risk and flood preparedness

Some research has shown that the public tend to define risk more broadly than experts, and take into account some of the societal implications of accepting the risks (McCarthy 2004). Differences in expert/lay perceptions of and responses to risk, including flood risk, often lead to increasing public mistrust of institutions and science (Cutter 1993; McCarthy 2004). People often use the ease with which examples of a hazard can be brought to mind as a cue for estimating probability of a hazard. As a result, experiences with hazards such as flooding should increase perceived risks. Moreover, the ability to recognize and read flood cues takes time and experience, and much of this local knowledge is reportedly being lost in today's more mobile societies (De Marchi *et al*. 2007).

In the recent past, there tended to be a perception that if people are aware of the risk they will 'correct' their perception and take appropriate action. This 'deficit model' of public understanding assumes that people lack accurate information and if this is provided their awareness deficit would be met (Wynne 1991). This assumes the public to be passive receivers of expert knowledge, rather than active citizens who evaluate multiple sources of knowledge to which they are exposed and who often have valid and useful lay knowledge. Even beyond the FRMRC case studies, the wider literature (Environment Agency 2007b; Burningham *et al*. 2008; Harries 2008; see also Chapter 18) shows that the key challenge in flood risk communication with the public lies in ensuring that those at risk engage with the issue of flooding at all. For example, in the FRMRC case studies with residents, in the 1 in 100-year return period flood risk areas where flooding has occurred in the recent past, levels of awareness of the local flood risk were variable. In response to a question on the extent to which their home was at risk from flooding, only in Chertsey did a majority 52% reply 'a great deal' or 'a fair amount'. In the other locations the proportions were much lower (32% in Glasgow and 41% in Trent). Emotional engagement with the flood risk also varied in the case studies, with nearly one-third indicating that they were not at all worried about the possibility of being flooded in the next 12 months. In recent surveys undertaken for the EA with 'at-risk' populations, less than two-thirds (Environment Agency 2004a, 2005a), and most recently only 52%, recognized that they lived in a flood risk area, with awareness varying from area to area (Pitt 2008).

Engagement with flood risk and the effectiveness of risk communication can be gauged by actions taken in advance to prepare for flooding. Indeed, professionals' efforts to raise awareness are intended to increase public preparedness for flooding, although it is likely that 'top-down' campaigns that aim to convince people that they are at risk are unlikely to succeed alone without engaging local stakeholders in raising awareness. A key preparatory action available to those at risk in England and Wales was to register on the EA's service, now called 'Floodline Warnings Direct', to receive automated telephone flood warnings. This service is now on an opt-out basis with automatic registration for those with listed telephone lines, with provision for de-registration and on-line registration planned for the future. According to the EA's Response to Flooding surveys on the flood in Carlisle in 2005 (e.g. Environment Agency 2005b), this system is now the main source of flood warnings. In 2008 about 350,000 households or businesses were registered on this system (Woolhouse 2008). However, given that there are approximately 2 million households at risk of flooding in England and Wales, it is clear that only a minority of households are covered to date. Surveys show that registration has been limited and variable; for example, in Carlisle in the 2005 flooding, one-third of those affected by the flood had signed up in advance for the service, while in the January 2007 flooding in the Midlands and North-East, 59% were registered (Environment Agency 2007a). Other surveys (Tunstall *et al.* 2007) have recorded lower proportions registered, and in the FRMRC case studies proportions were much lower, 11% in Trent and only 2% in Chertsey.

Taking out house-and-contents insurance or checking that policies cover flooding are other preparatory actions commonly taken by those at risk in England and Wales (Environment Agency 2005a; Tunstall *et al.* 2007). The FRMRC case studies and other research (Tunstall *et al.* 2007) show that more substantial preventative measures (such as structural adaptations to property including buying floodgates, installing pumps, raising floors or building walls) were undertaken by small minorities. Although more common than structural adaptations, only a minority made behavioural adaptations such as changing furnishings and relocating household items.

A flood event in itself can be regarded as a significant communicator of risk. Kates (1962) in his seminal work, *Hazard and Choice Perception in Flood Plain Management*, found a link between experience and awareness of flood risk. Subsequent research and recent studies have confirmed experience of flooding as the key factor in flood awareness and preparedness (Environment Agency 2004a; Tunstall *et al.* 2007). In the FRMRC case studies, perceptions of flood risk and worry about the possibility of flooding in the next 12 months (and readiness to undertake preventative measures) were higher for those who had been flooded at their property. Some significant social factors affecting levels of flood risk awareness and the propensity to take action to minimize flood impacts have been identified in English research (Fielding *et al.* 2005). Regression analysis showed a range of socioeconomic indicators to be significant in increasing flood awareness. As before, previous flood experience emerged again as by far the biggest factor. Those who had been resident for 1 year or less showed markedly less awareness than longer-term residents.

In an alternative approach, psychologists have attempted to explain responsiveness to flood risk communications and people's predispositions to act to reduce their flood risk. These attempt to improve risk communications by matching message types to individual psychological traits. The audience segmentation methods common in social marketing are based on this idea (e.g. O'Neill 2004), but to date these have not proved altogether successful and useful in the UK.

Flood warning communications

It can be argued that flood warnings are the most significant form of flood risk communication that takes place between professionals and the public, given that warnings have the potential to reduce the impact of flooding, including the risk to life, health and stress effects, disruption to life and damage to property.

Yet it is important to recognize that flood warnings in the UK and elsewhere often fail to reach those at risk. As research shows, success in dissemination is very variable by event and area. For example, in Carlisle and the Northwest of England in 2005, 46% (Environment Agency 2005b) recalled receiving a warning; in later events in January 2007 (Environment Agency 2007a) the proportion was 64%; in events in 2003 and 2004 (Environment Agency 2005c) the proportion was 33% overall. Variable success in warning dissemination may in part be accounted for by the nature of the flood events and the difficulty in predicting and providing advance warning of some extreme rainfall events. Such events include the Boscastle event in Cornwall (Environment Agency 2004b), extreme rainfall events in rapid-response catchments, extreme events occurring outside areas with a flood warning service, and events mainly involving surface water runoff and drainage systems overwhelmed by heavy and prolonged rain, as in Hull in 2007 (Pitt 2008).

The social characteristics, experience and attitudes of recipients may also be factors in warning dissemination. Most obviously, in England and Wales, being registered to receive an automatic telephone warning has been shown to be a key factor, and registration in itself has, not surprisingly, been found to be associated with flood experience (Tunstall *et al*. 2005). In some studies, in Carlisle, for example, those with certain social characteristics were found to be more likely to recall receiving a warning: those aged over 35; households with a resident aged 65 and over; those owning rather than renting property; those aware that their property was at risk from flooding; and those who had undertaken preparatory action of some kind (Environment Agency 2005a). Past experience and awareness of flooding and long-term residence are other factors that have been associated with warning receipt (Tunstall *et al*. 2005).

The factors that constrain and enhance responses to flood warnings have attracted substantial research interest in the UK and elsewhere (e.g. Drabek 1986, 2000; Environment Agency 2007b; Parker *et al.*2009). The message itself is important, here involving the translation of flood risk science into a message suitable for a non-professional audience. Drabek (1986, 2000) has outlined message characteristics that have been validated repeatedly in the research literature as enhancing response to hazard warnings. The research shows that warnings are more likely to be believed and acted on if they have these characteristics:

• Clarity (and lack of ambiguity).[5]
• Precision and detail as regards location, timing and magnitude in message content (Gruntfest 1977; Mileti and Fitzpatrick 1992).[6]
• Consistency in message content and across media (e.g. Mileti and Fitzpatrick 1992).
• Multiple messages and repetitions (Baron *et al*. 1988).
• Allow a means of confirmation (in the sense that it is recognized widely that initial behaviour on receipt of a warning is to seek confirmation).
• A credible, known and trusted source of warning is perhaps the most important characteristic. This may be an official source, but hazard warning messages delivered by family and friends have also been found to be effective (Dow and Cutter 1998).

In England and Wales, the EA has sought to provide for all these characteristics in its warning communications. In particular, the EA has sought to make itself known to the public as a warning agency and to build up public trust. It has had some success over the years in increasing spontaneous awareness of its responsibility for issuing flood warnings in flood risk areas (Environment Agency 2005a). However, in the FRMRC case studies a wide range of organizations were cited as preferred warning agencies. In only one area, Chertsey was the official agency (the EA) the agency most commonly chosen as preferred and most trusted to give a reliable warning. Some even mentioned the 'National Rivers Authority', which the EA superseded in 1996, indicating how long it takes for an agency to become known and recognized by the public.

Although technology is a key driver facilitating change in FRM, other contextual issues in turn

[5] Ambiguity creates a tension, despite the ethical appropriateness of communicating uncertainty in messages when it exists (see Faulkner *et al*. 2007).

[6] Again this aspiration is at tension with the scientists' awareness that the message is always to some degree uncertain.

facilitate the use of that technology in decision-making. Resources, expertise, knowledge, professional relationships and time are a few of the key constraints that can act on the ability to utilize the technology usefully. If such issues are not addressed then it could be argued that decisions informed by advances in technology may not be timely or actionable. Furthermore, the vulnerability of technology in times of flooding, the dangers of over-reliance on communications technologies, and the issue of access to and acceptability of these technologies for some members of the public must be recognized. Indeed, the importance of understanding the requirements of both the professional and non-professional recipients of communications and the constraints acting in FRM was underlined by the FRMRC research.

The media

The media appear increasingly to play a key role in enhancing awareness in general. Local broadcast media, both radio and TV, have a very significant role as a means of disseminating flood warnings before an event, and in flood risk communications with the public during and immediately after flood events. For example, during the 2007 flood event in Gloucester, a daily live locally broadcast press conference given by senior emergency managers provided a main mechanism for keeping the public informed about the event. Local radio websites provide a further source of information to the public to supplement the EA's website. For example, the BBC Radio Gloucester website recorded 7.9 million hits in 2 weeks during the summer flood event of 2007 affecting Gloucester and surrounding area, indicating a huge appetite for information, and not just from local people (Cameron 2008). Local BBC radio stations and their associated websites not only transmit detailed information originated by professionals on flood events but also actively engage with their audience by broadcasting telephone calls from local people and allowing local people to blog on their websites. Thus local media broadcasting about flood events has become a two-way interactive process with the audience. At national level, the EA regards the commitment of the BBC to broadcast flood warnings on national and regional BBC weather forecasts (and to highlight the Floodline automated call number for people to ring for further information) as a major benefit in raising awareness of flood risk generally and among those at risk in a particular event.

For the Carlisle floods of 2005, post-flood debriefing reports reveal that the work of one radio station in particular held together the emergency response when other communications mechanisms failed. The emergency services reported that they gained valuable information from this radio station, and this was then used to good effect in the emergency response to the public. This shows that the role of local radio in communications during a flood event can extend beyond that of informing the public and can be of importance to professionals as well (Government Office for the North West 2005). Furthermore, during major flood events, newspaper and broadcast coverage of the event and its aftermath generates widespread interest and debate among policy-makers, professionals concerned with FRM and the public (Johnson *et al.*2005). The point made by Johnson *et al.* (2005) was made even more valid after the 2007 floods in England. Interest may continue for a year or more as policy-makers follow up stories about the event and absorb and act upon lessons learned. In the UK this is exemplified by the Pitt review (Pitt 2008) and the earlier Bye report (Bye and Horner 1998). Thus policy change may be accelerated when major flooding occurs (Johnson *et al.* 2005).

Summary

This chapter has explored the challenge of effective communication in contemporary FRM, concentrating on the situation in the UK. It has been emphasized that FRM has moved away from the engineering responsibilities of the past to one where risk communication has a much more central role, and this means talking more with the public at all stages of the hazard cycle. In the UK, improvements in science's ability to anticipate events using improved radar and modelling systems all suggest that new communication tools will emerge soon in the professionals' toolkits, frequently seen as animated visualizations, to help at the public interface in reducing flood risk. The expanding field of flood risk science, therefore, offers great potential for the future.

Professional settings are changing fast, and we have explored how and where professionals need to adopt the translation of science in both intra- and extra-professional communications. As science now increasingly struggles to validate models of a non-stationary future, flood risk professionals are also challenged to communicate the level of uncertainty in their risk communications. There are clearly ethical obligations do so, yet Pitt (2008) emphasized that people are often confused over different pieces of information that they receive, which suggests the need for a single definitive set of advice and information on flood prevention and mitigation. These two observations are in tension.

We have demonstrated that the physical characteristics of the flood threat and the social and institutional context differentiate professional choices about risk communication strategies with the public. We have also emphasized that public societies and communities are complex and diverse, so that individuals, groups and organizations within them will construct risk, and understand and respond to risk communications in different ways. Chapter 18 emphasizes that a key requirement for professionals seeking to communicate with the public is an understanding of the experiences, attitudes, values and needs of those to whom communications are addressed, through dialogue, engagement and social research. This includes an awareness that the psychological characteristics of the recipients of risk communications are important factors underpinning the messages and the language in which they are couched.

We conclude from existing research and the specific findings of the FRMRC's phase 1 activities that to develop effective flood risk communication strategies at the public interface requires an improved understanding of factors that influence flood awareness and of how the risk of flooding is constructed by those at risk (Burningham *et al*. 2008). Risk is neither a process that is simply attributed to natural processes (e.g. hazard) nor an objectively given constant. Rather people's understandings of flood risk are the result of a process of social construction (i.e. norms, values and belief systems) and not simply of perception and information (Steinfuhrer and Kuhlicke 2007).

New methods to assess and model floods are becoming available from the expanding field of flood risk science. Meanwhile, the use and potential these tools offer as risk communication tools is being revisited, although as this chapter has revealed, the message is not always getting seamlessly through to the public interface. However, awareness and warning about flooding is being particularly enhanced by the media, which have an increasingly vital role to play.

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20 Socio-Psychological Dimensions of Flood Risk Management

SUE TAPSELL

Introduction

Floodplains are among the most densely populated areas in the world (Kron 2002). As the risks of flooding increase through a changing climate and often unpredictable weather patterns (IPCC 2007; Evans *et al.* 2008), no protection works can guarantee future security. This recognition has resulted in the recent shift in policy from flood defence to flood risk management (e.g. Defra 2005). Added to this is the increased risk from groundwater flooding or intense pluvial flooding where the capacity of drainage systems may be insufficient to deal with the volumes of water involved. Environmental changes are also taking place against a backdrop of wider societal change that may alter the likelihood of human exposure to hazards, as well as people's susceptibility to their impacts (Few 2007). Continuing development within floodplains, increasing population densities and mobility, accumulation of household goods, and little awareness of flood risk are all contributing factors to such exposure and susceptibility.

Flood events, like other natural disasters, can have varying and significant impacts upon those who are exposed to them, as well as those who have to respond to such events. In 2007 alone there were 200 major floods worldwide, resulting in more than 8000 deaths and affecting 180 million people (Pitt 2008, p. 15). Apart from loss of life and serious injury, flood events may also impact upon other aspects of human health and well-being and upon social relations, as well as causing extensive damage to properties, infrastructure and the natural environment. In the past, the intangible socio-psychological aspects of flooding were largely ignored both in policy terms and in practice. Technological solutions to flood risk were emphasized and impact analysis tended to focus on economic and financial damage and losses (Brown and Damery 2002). In addition, the response to hazards de\monstrated a 'command and control' mentality that focused on clean-up and the rescue of survivors. However, recent years have seen an increased recognition of the social aspects of flooding and in particular flood impacts upon people as **receptors** (e.g. Bye and Horner, 1998; Mileti, 1999; Evans *et al.* 2004; Pitt, 2008).

There is now growing concern regarding the longer-term impacts of climate change, including flooding, on human health and well-being (WHO 2002; IPCC 2007). Dramatic media images of flood events from across the globe such as the Asian tsunami (in 2004), Hurricane Katrina (in 2005), the summer floods in England and Wales (in 2007) and exceptional flooding in Cumbria (in 2009) have helped to focus attention on the human aspects of natural hazards. Once floodwaters have receded and the media coverage has ceased there has often been a perception that the event is over. However, for those affected by flooding in their homes and businesses the majority of their problems are just beginning, and people have to cope with the significant aftermath of the flood,

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not only at a practical level but also socially and psychologically.

With the growing emphasis on flood risk management (FRM) strategies there is an increasing recognition by flood risk policy-makers and managers of the need to include more consideration of the socio-psychological aspects of floods. Recommendations from the Pitt Review (Pitt 2008) suggest the need to begin by assessing the needs of affected communities. Similarly in the USA, events surrounding Hurricane Katrina and its aftermath have increased awareness of the importance of considering influences beyond national economic development and have led to calls for fully integrating other social effects into project analysis and decision-processes (Deeming and Durden 2008). According to Dunning (2009, p. 7): 'One of the lessons of Katrina and [Hurricane] Rita has been that of all the social effects associated with storms and floods their impact on socially vulnerable populations has been woefully overlooked and underestimated.' Dunning and Durden (2008) suggest tools and methods for developing information on social factors and a framework for using these in the planning process. In the UK, the Department of Health (DH Emergency Preparedness Division 2009) has also recently developed guidance on psychosocial care for people following disasters, based upon work undertaken for NATO and the EU (Williams *et al.* 2009). These reports conclude that the way in which people's psychosocial responses to disasters are managed may be the defining factor in the ability of communities to recover.

The above discussion suggests the requirement of including more socio-psychological dimensions when developing FRM policy and practice. This is essential with the move to more non-structural approaches to managing flood risk, particularly those that require specific behavioural responses from the public and other stakeholders. This chapter will focus on the socio-psychological dimensions of FRM. It will begin by outlining a conceptual framework for analysing these aspects, followed by a discussion of some of the key influencing factors. A better understanding of these socio-psychological aspects can aid flood risk managers and other responsible agencies in more effective planning, in seeking more effective measures to prevent or mitigate these impacts, in promoting speedier recovery and more effective response, and in developing strategies to increase resilience and capacity to cope with future flooding. Discussion will be confined to evidence from developed countries.

A Framework for Analysing the Socio-Psychological Dimensions of Flood Risk Management

The socio-psychological dimensions of FRM can be defined as those aspects that have potential to adversely impact on the social, psychological and physical well-being of those affected; in other words, those aspects affecting a person's social and psychological functioning. Floods and decisions around FRM have the potential to seriously impact upon this functioning, often with long-term consequences for individuals and communities. These impacts affect individuals (through mental processes and impacts) and their interactions with others (social structure and relations) (Cote and Levine 2002).

A simple conceptual framework has been developed to aid the analysis of the socio-psychological dimensions of FRM (Fig. 20.1). The framework is based on the 'hazard' or 'disaster management cycle' often cited in disaster management literature (e.g. Wisner and Adams 2002; Few 2006). Risk management deals with the preconditions, causes and impacts of hazards (Rohrmann 2003). Its multiple tasks need to be implemented before, during and after an emergency or disaster. Preparedness, damage control, recovery and mitigation are crucial aims of risk managers. These tasks require administrative, technological, medical and socio-psychological means and resources. The disaster management cycle thus divides disaster events into various stages, normally: before, during and after, or pre-onset, onset and post-onset phases. In the framework presented in this chapter four phases are presented: preparedness, emergency response,

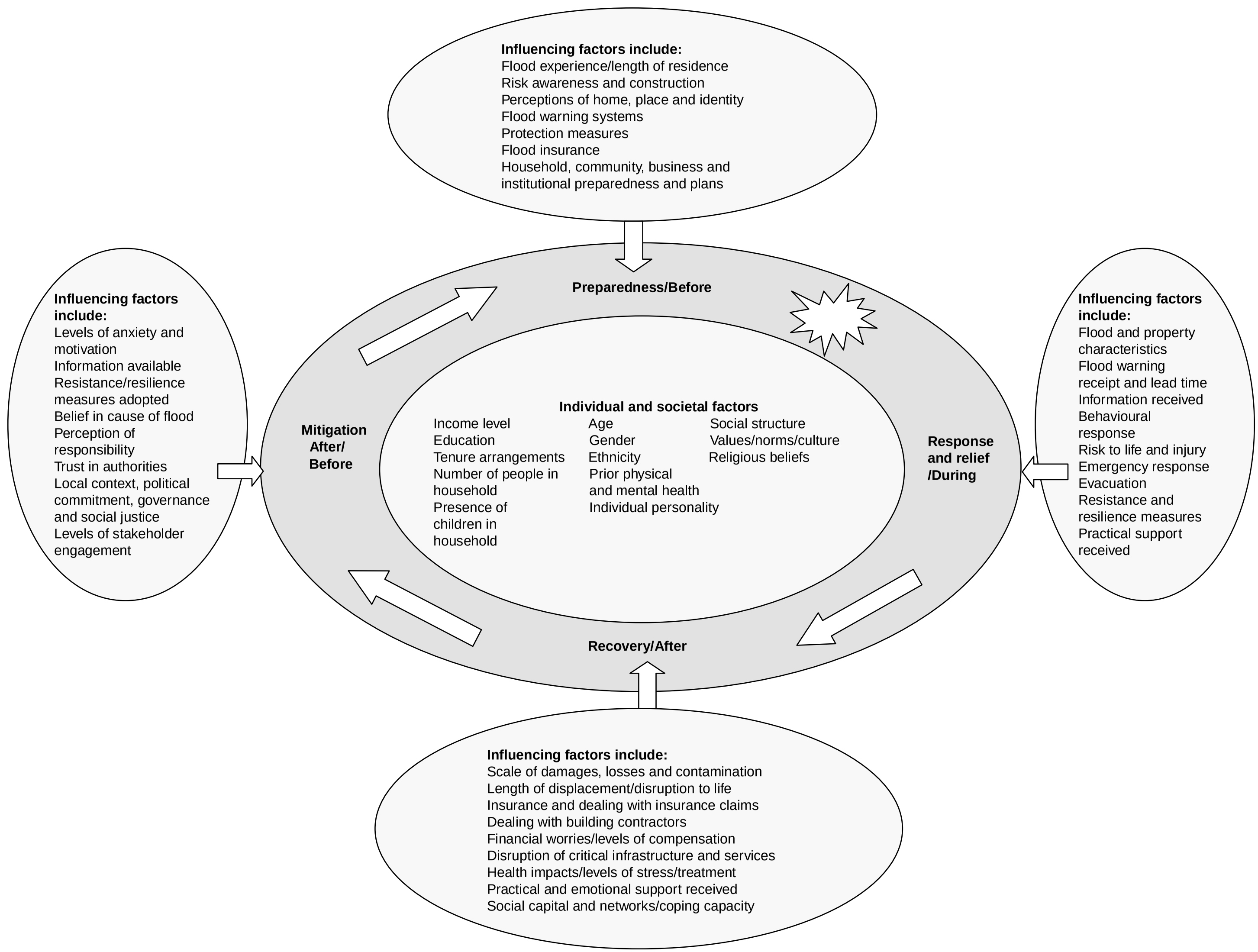

Fig. 20.1 Framework for analysing socio-psychological dimensions of flood risk management.

recovery and mitigation. This temporal aspect of the cycle is important; however, flood impacts and responses are dynamic and may often be overlapping or present during more than one phase. It should therefore be noted that the various influencing or intervening factors reported here are not necessarily discrete to one individual phase. At the centre of the cycle are the individual and societal factors, which may impact upon the various socio-psychological dimensions of floods at each stage of the disaster cycle.

Although being useful as an analytical tool, the disaster management cycle has been criticized for portraying disaster response in a circular fashion, which is said to reinforce the perception of disasters as an aberration from normal conditions, and that conditions will return to normal once the event has passed (White *et al.* 2004; Few 2007). This assumes, for example, that certain conditions, such as social vulnerability, are not pre-existing in normal circumstances within affected societies, which is rarely the case. In reality, pre-flood conditions such as poverty and vulnerability may be simply recreated following flooding. The disaster cycle also fails to acknowledge that, in certain circumstances, and particularly in conditions of poverty, losses many lead to increased vulnerability, making people **more** susceptible to future flooding. White *et al.* (2004) refer to this as a negative downward disaster 'spiral' rather than a cycle. However, the disaster cycle also implies that lessons will be learned and that positive change will take place. In this sense White *et al.* (2004) promote the concept of the 'virtuous spiral' of risk reduction whereby lessons can be learned from a disaster that may result in positive adaptation and outcomes. Although acknowledging the validity of the critical arguments cited above, this chapter suggests that positive adaptation is possible.

Given the necessary political commitment, FRM can be improved to reduce or mitigate adverse socio-psychological dimensions of flood events and enhance positive ones. The framework shown in Figure 20.1 can be useful in highlighting the different factors that may influence socio-psychological aspects of floods during the different phases of the disaster cycle, indicating where and when to target resources and inputs, and aiding policy-making and planning for future flooding. Each stage of the disaster management cycle will now be discussed in turn and key issues highlighted.

Preparation and Planning Before Flooding

Risk awareness, construction and flood experience

It is generally acknowledged that preparation and planning for natural hazards such as floods can help to avoid or reduce damage and losses and thus lessen many negative socio-psychological impacts. There are many ways in which people can prepare and plan for floods; with the increasing risk of flooding these measures become even more important. Key factors in preparedness and planning for floods are: flood experience (often related to length of residence in an area), awareness and acceptance of the risk, and a desire and ability to take mitigating actions. Awareness is often related to past experience of flooding, which has been shown to be even more significant in influencing flood preparedness than simply raising awareness. It is particularly difficult to raise awareness where no history of flooding exists, and this has implications for areas with a low probability of flooding, but where the potential consequences could be high (Shaw *et al.* 2005).

Research by Burningham *et al.* (2008) found the following factors to have an important effect on flood risk awareness: flood experience, length of time at present address, tenure, age and social class – with social class being the most significant in predicting awareness. New residents to areas appeared particularly unaware of flood risk, which indicates a need to include information on flood history to potential purchasers of properties. Tenure has been shown to be a factor in risk awareness in the UK (Tunstall *et al.* 2006); homeowners in particular may seek out information on flood risk as the home represents a significant financial investment. Renters are often unaware of such risk as landlords rarely take responsibility for informing their tenants. Other factors suggested as being significant in affecting flood risk awareness include: the nature of flood events, local flood history and institutional factors such as awareness campaigns, social networks and community preparedness.

Although there is evidence in England and Wales of a general increase in flood risk awareness since 2000 (Defra 2005), very low levels of awareness were reported following the extensive summer 2007 floods (Pitt 2008). When people personally take risk-reducing measures it implies that they are both aware of the risk of being flooded **and** that they attribute certain significance to these measures (Steinführer *et al.* 2007). Evidence from England (Pitt 2007; GfK NOP 2007; Norwich Union 2008) and wider Europe (De Marchi *et al.* 2007; Steinführer and Kuhlicke 2007) shows that even though people may be aware of flood risk this does not mean that they take actions to prepare themselves, and that relatively few people take effective individual damage avoidance measures on receipt of a flood warning.

There is often a tendency for people to deny personal flood risk. Although many perceive their local area to be generally at risk, people do not necessarily translate that risk to their own property (Steinführer and Kuhlicke 2007; Burningham *et al.* 2008). All of these perceptions and behaviours are related to people's social constructions and evaluation of the risk (Steinführer *et al.* 2007). Shaw *et al.* (2005) indicate that it is those awareness campaigns that reflect social values and perspectives that are likely to be the most effective. Renn (2008) also highlights the closeness in the connection between knowledge and values; the stage at which risk is framed and defined will inevitably involve social values in determining what risks are socially significant and the setting of goals. The 'information-deficit' model widely used by flood risk managers in the past is said to neglect the socially embedded and contextualized manner in which people make sense of the world. Risks need to be viewed in the context of evaluations of local life and the local environment (Burningham *et al.* 2008). People must also be motivated to take preventive actions that result in decreased personal risk and losses. There is therefore a need to increase awareness not just of the probability of floods but also of the negative **consequences** upon households and communities, including the length of the recovery process. The role of risk construction, attitudes and perceptions in individual decision-making behaviour needs to be better understood, as well as how such a risk behaviour framework may be used as a tool to formulate effective response strategies. More effective flood risk communication strategies will also need to be developed; see Chapter 19 for a discussion on risk communication.

Perceptions of home, place and identity

The importance of emotions for risk perception is beginning to emerge (e.g. Slovic *et al.* 2002) but their impact on risk preparedness and response has as yet received little attention. Emotions are particularly important when it comes to people's perceptions of home. Flooding is said to undermine such perceptions and people's individual sense of self and place identity (Fullilove 1996; Tapsell *et al.* 1999; Cox and Holmes 2000; Tapsell and Tunstall 2008). Psychological attachment to the home can also be a factor in risk denial (Sime 1997; McCarthy 2004). People have a strong emotional attachment to their homes, can experience severe distress when they are damaged, and have reported feeling less attached to their homes as a consequence of flooding (Tapsell *et al.* 1999). Security in an area as a place to live may also be lost, thus a place that was once familiar can suddenly become unfamiliar and fearful following flooding, changing people's relationship with place (Tapsell and Tunstall 2008).

> Your home is your haven, you go there when you've had a bad day. Suddenly it's not there anymore.
>
> **Resident, Gloucester (GfK NOP 2007)**

Previous flood experience (and the appreciation of the impact that this can have on the home and people's identity) has thus been shown to be more significant in preparing for flooding than simply awareness.

Preparedness actions and measures

Those with prior experience of flooding inside their home have been shown to be more active in taking certain preparedness measures, and significantly more of those who had been flooded three or more times have taken more drastic preventative measures (Tunstall *et al.* 2007). Preparedness actions taken by households and businesses in the UK have ranged from keeping alert for flood warnings during high-risk months, not keeping irreplaceable items on ground floors and acquiring sandbags, to moving valuables, personal property and cars to safety (Tunstall *et al.* 2007). However, Harries (2008) argues that one reason why people do not prepare for flooding is because such measures are perceived as endangering other needs that are more immediate and pressing, such as protecting people's emotional security and their existing representations of security (e.g. the home being a safe place), which may result in denial of being at risk. For others, flood mitigation measures such as flood gates were rejected as they lessen the visual conformity of their homes to an idealized norm, and are often perceived to reduce the value of properties by alerting potential buyers to the flood risk.

Signing up to receive flood warnings is one action that people can take to be prepared for flooding. Being aware of flood risk and/or experience of flooding have been highlighted as key factors in whether people adopt flood warning technologies (e.g. Tapsell *et al.* 2004; Fielding *et al.* 2006). One assumption is that if people are aware of and accept the risk, they are much more likely to be receptive to flood warnings. Multimedia flood warning dissemination systems now enable more people to be contacted and also allow people more choice in the various media via which warnings can be received. However, in 2007 only 20% of people in regions flooded in England had registered to receive flood warnings when invited, and the figure rises to only around 41% for England and Wales as a whole (Pitt 2008, pp. 319–320), reinforcing the idea that factors other than awareness may be important influences.

Taking out insurance is a common form of preparedness measure by residents in flood risk areas. However, lack of risk awareness may mean that people do not specifically check if their policies cover them for flooding. The key factor often seen as important in insurance take-up is social grade (i.e. income levels, education and awareness) with those in the lowest social groups and those living in 'vulnerable' housing significantly less likely to have insurance cover (Tunstall *et al.* 2007). Past experience of flooding and home ownership can also be important factors in decisions to purchase flood insurance. Tenure is important in that those not owning or buying their property are less likely to have insurance of all kinds (Tunstall *et al.* 2007), but this has been found to be dependent upon national tenure cultures; for example, tenure is important in the UK but not generally so in Germany (Steinführer and Kuhlicke 2007). However, most of the actions reported above require individuals or businesses to take the initiative before a flood event; to date there have been few institutional pressures to encourage this.

Institutional measures in the form of emergency plans for flooding, provision of evacuation centres and temporary flood defences are other means of preparedness along with business continuity and contingency plans. Evidence from the summer 2007 floods indicates that many organizations and businesses still do not prepare such plans (ABI 2007). Individual household and community flood plans also have the potential for increasing preparedness but as yet have been little researched. Flood maps may be one way of raising awareness of flood risk. However, these maps refer to fluvial and tidal flooding and not to the increasing risk from pluvial or groundwater floods. Community flood warden schemes can be an effective preparedness measure, although wardens have been found to be difficult to recruit and sustain. Recent evidence from England has shown that such schemes can be successful but they rely on good community engagement and involvement (Environment Agency, personal communication, 2007).

Response and Relief During Flooding: Damage Control

Figure 20.1 outlines many of the factors that may influence response and damage control during flood events and thus impact upon socio-psychological dimensions of FRM. Some of these factors will not be discussed except to highlight their impact upon resulting damages and losses, for example, depth and velocity of floodwaters and sediment/debris content and contaminants. Those living in single-storey properties may lose their entire home contents and suffer damage to every room, while for those in multi-storey properties damage is generally confined to ground and lower ground living space. However, having the main living areas flooded, particularly kitchens, creates huge disruption and distress (Tunstall *et al.* 2006, 2007; Werritty *et al.* 2007). The receipt of adequate warnings and timely support can often reduce potential flood losses and distress, particularly for people who are unable to move heavy belongings. Lack of practical support has often been raised as a criticism of response agencies (Tapsell *et al.* 1999; Pitt 2008).

Flood warnings and behavioural response

As highlighted above, even if effective warning systems are in place there is still uncertainty over if or how recipients will respond upon receipt of a warning. Research has shown a preference for face-to-face warnings, for instance from flood wardens and door knocking, and also that a large proportion of the population still do not have access to the internet to receive information (Tapsell *et al.* 2004). Age and receipt of a flood warning have been shown to be significant factors affecting ability to take actions (Parker *et al.* 2007). Flood warnings in this instance were a significant driver of behaviour before and during a flood event. However, actions taken by recipients are often ineffective; for example, many people try to prevent floodwaters from entering properties, which reduces time that could be spent saving belongings.

> I threw bath towels down. That was the first thing I can remember doing. Grabbing bath towels and throwing them to the bathroom door as I could hear it coming up through the toilet. . .
>
> **Resident, Kidlington, 1998 (Tapsell *et al.* 1999, p. 14).**

Flood warning lead time is also important in determining what actions people can take to reduce impacts. Telephone warning systems work best for slow rising river floods but may not be appropriate for floods in rapid-response catchments or in the case of intense pluvial flooding. Flood warning methods need to be tailored according to the type of flood and to recipient and location characteristics. Crucially they need to be focused on facilitating **effective responses** during floods rather than focusing solely on warning large numbers of people (Fielding *et al.* 2006; Twigger-Ross *et al.* 2008). The focus on meeting performance targets for recruitment to warnings systems means that considerable resources are targeted at that activity, which makes taking a 'response focus' to warnings harder (Twigger-Ross *et al.* 2008). These findings are significant and indicate the need for institutional change.

Risk to life

Socio-psychological dimensions to FRM can also include those associated with people's health during a flood event. Although the threat to life and injury from floods tends to be much greater in developing countries and in countries like the USA where the scale of flooding is greater, there can still be significant risk. Coastal flooding in particular has the potential to pose greater risk to life than river flooding (Baxter *et al.* 2001). The August 2002 floods in Central Europe resulted in more than 100 fatalities (WHO 2002). Yet to date, we know little about the specific causes of death from floods (Jonkman and Vrijling 2008) although recent research has attempted to analyse this for Europe (Priest *et al.* 2007). Several methods have now been developed as means to calculate this potential risk to life in both quantitative and qualitative terms (for a review see Priest *et al.* 2007; Jonkman and Vrijling 2008). Mortality associated with a flood depends not only on the flood characteristics (e.g. depth, velocity and speed of onset) but also on the way people respond to floods (Priest *et al.* 2007). Deaths are strongly related to risk-taking behaviour, particularly among males (Jonkman and Vrijling 2008), and the World Health Organization (WHO 2002) estimates that up to 40% of health impacts due to flooding result from such behaviour. A high percentage of deaths are vehicle-related, and males are almost twice as likely as females to be killed, with young people particularly at risk. This highlights the importance of flood risk education, as many people are ignorant of the power of floodwaters. Recent research in the UK has called for questions on the risk of driving through floodwaters to be included in driving test examinations (Cave *et al.* 2008).

Evacuation before floods can reduce risk to life in the case of severe flooding and is frequently necessary post-flooding as properties will often be uninhabitable for many months. However, poorly organized and managed emergency response and evacuation can add to the distress to those who are flooded, as evidenced during Hurricane Katrina in the USA (Nossiter and Schwartz 2008). The 2007 UK floods also highlighted the need to re-evaluate the location of evacuation centres, as many were themselves flooded (Pitt 2008). Although all of the above aspects of FRM can have significant socio-psychological dimensions and impacts upon those affected by floods, the majority of such impacts have been reported once the floodwaters have receded, during the recovery process.

Recovery after Flood

Post-Flood disruption and financial concerns

Recovery following flooding is dependent upon a number of factors, key to which are the extent of damages and losses, and individual, household and community resources available to deal with these. The extent of damages will usually determine the length of displacement and disruption to life. Where damage is extensive many people have to live in temporary housing such as hotels, mobile homes and rented accommodation for many months, and lengths of up to a year are not uncommon. Such displacement can have significant socio-psychological impacts. Post-flood disruption to life has been reported in the UK as the most significant stressor affecting people's health and well-being (Green *et al.* 1985; Tapsell and Tunstall 2001; Carroll *et al.* 2006). Psychological distress is often more reflective of the difficulties and hardships encountered during recovery than the impact phase of an event.

> The world was on its head, wasn't it? Everything you knew to be normal didn't exist any more. Nothing was right was it?
>
> **Resident, Chesterfield (GfK NOP 2007, p. 33).**

Moving back to properties before they are adequately dried and aired can cause further distress; people in Carlisle were still reporting dampness and problems 3 years following the 2005 floods (Fernández-Bilbao *et al.* 2008). There is thus a need for advice on when to reoccupy damp properties and for definitive guidance on best practice in drying properties.

Disruption to critical infrastructure and services (e.g. water supply) following flooding, and poor provision of logistical support, can result in additional distress for those affected, and even for those not flooded (Pitt 2008), including increasing the risk to public health. Contingency planning for the loss of such services is crucial, and better planning is also needed to source essential supplies in major emergencies (Water UK 2008). Further issues during recovery relate to dealing with insurance claims and contractors repairing and restoring properties. A large-scale study in England and Wales showed that dealing with insurance claims was statistically the most significant factor affecting people's post-flood psychological health (RPA *et al*. 2004; Tunstall *et al*. 2006). Evidence from 2007 indicates that some UK insurance companies have improved their service compared with earlier floods (Pitt 2008). However, lack of insurance take-up among low-income households is still an issue that needs to be addressed. Such lack of insurance can result in people having to return to homes that have not been adequately dried and repaired, or a delay in their return due to lack of financial resources.

Loss of livelihood following flooding can be particularly stressful for those affected, particularly where social welfare systems or social networks are weak. Taking time off work to deal with recovery can lead to loss of wages and significant financial concerns for those on low incomes. Increased debt enquiries were reported following flooding in Hull, and mortgage and loan arrears increased along with arrears in bill payments (Pitt 2008). Farmers often bear the brunt of impacts in rural areas, from loss of livestock and from uninsured losses, for example to crops (GfK NOP 2007). Other businesses may also face direct and indirect losses or reduction in trade. Owners of small businesses in Carlisle spoke of the financial and practical difficulties of recovery with little or no support available, and stressed the psychological impacts as well as the financial (Fernández-Bilbao *et al*. 2008).

Taylor (2000) suggests that governance structures and agencies need to become more responsive to community needs. Following the 2005 Carlisle flood, a partnership project, Communities Reunited, was set up to support the affected communities; lead agencies included churches, local authorities and voluntary agencies. This initiative was seen as very successful in meeting community needs and in aiding recovery. Residents spoke of needing very practical support, and someone who would listen and who would help find a path through the maze of decisions that had to be taken (Fernández-Bilbao *et al*. 2008). Importantly, Carlisle authorities have used the recovery process as an opportunity to regenerate parts of the city, rather than simply returning them to their pre-flood state.

The impacts on human physical health from floods

There is presently a weak evidence base to assess the health impacts of flooding (WHO 2003; Hajat *et al*. 2005). Relatively few rigorous studies have been undertaken, and it is extremely difficult to assess the duration of symptoms and disease, as well as the attribution of cause, without longitudinal data. Few and Matties (2006) presented findings from a wide-ranging epidemiological review of the evidence base for health outcomes from flooding and a review of literature analysing mechanisms of response to such health risks. The greatest burden of mortality from floods is from drowning, heart attacks, hypothermia, trauma and vehicle-related accidents. More frequently, common physical health effects result from minor injuries (Schmidt *et al*. 1993; Manuel 2006), diarrhoeal episodes (Wade *et al*. 2004; Reacher *et al*. 2004) and respiratory disease (Menne 1999; Franklin *et al*. 2000). Skin irritations, burns, electrocutions, and chemical and carbon monoxide poisoning are also common. The effect of floods increasing the risk to public health is relatively rare in developed countries due to good sanitation and water supplies (Malilay 1997; Meusel and Kirch 2005; Ahern and Kovats 2006), although it is not unknown (see above and Chapter 21). However, with normal routines disrupted other aspects of life may also suffer; diet and exercise is one such area. Nutritional status can be affected due to reliance on different and poorer quality foodstuffs. People may have to rely on takeaway foods as they have no kitchens in which to cook, as well as often a lack of motivation to cook.

> We had a takeaway every night for three months and I don't mind admitting that...what me and my wife did we had one meal a day because you couldn't go for a takeaway in the morning, in the dinner time and so what we did was and it's daft because we've got used to it, we eat one meal a day, the baby gets fed at crèche.
>
> **Resident, Barnsley (GfK NOP 2007, p. 13)**

A further social effect of floods is due to the likely disruption of normal health care provision and social programmes (Ohl and Tapsell 2001; Meusel and Kirch 2005). Following Hurricane Katrina it has been estimated that some diabetics went as long as 6 months without insulin (Berggren and Curiel 2006). Clear and consistent advice is necessary on both physical and mental health risks and needs to be widely available (Health Protection Agency, personal communication, 2008).

Mental health impacts from floods

In recent years the specific mental health impacts of flooding have become increasingly recognized as significant in affecting people's well-being (Tapsell and Tunstall 2006; McNamara 2007). There is strong evidence that flooding can have an adverse effect on common disorders such as anxiety and depressive illness. Reacher *et al*. (2004) and Tunstall *et al*. (2006) report significantly higher rates of psychological impairment among those flooded compared with those at risk. Those who are diagnosed with psychiatric problems are also more likely to have a greater number of physical health problems than those who are not diagnosed (Stoudemire 1995). Mack *et al*. (2007) reported that 44% of children in New Orleans showed symptoms of new mental health problems such as depression, anxiety and sleep disorders, although research into children's psychological responses to disasters and emergencies is still at an early stage (Evans and Oehler-Stinnett 2006). Moreover, there is growing evidence that disaster victims may continue to experience psychological health symptoms long after the event (Preiss *et al*. 2004; Manuel 2006; Tunstall *et al*. 2006).

Other chronic problems identified by Norris *et al*. (2001) include troubled family and interpersonal relationships. Following the 2007 UK floods, 22% of people surveyed who were married or living with partners reported an effect on their relationships, with those forced to move out of their homes almost twice as likely to report problems (Pitt 2008, p. 361). In Carlisle, loss of motivation to pursue personal interests and hobbies following flooding was reported, when in fact such activities would probably have aided people's recovery (Fernández-Bilbao *et al*. 2008). The keeping up of routine social activities is important, as these organized activities help to maintain social networks and support camaraderie. A further suggestion is for people to take 'pamper days' to aid their recovery. Finally, disasters can have an impact on front-line workers, on media personnel and on the extended families of those affected (Pitt 2008; Whittle *et al*. 2009); this aspect of flooding has been poorly researched to date.

Socio-psychological resources and support

During and following disasters individuals and communities may respond to the threat by mobilizing personal and social resources. An individual's capacity to come to terms with a traumatic experience is greatly influenced by his or her social context. Secure, supportive relationships are essential for people's communication and processing of the traumatic experience and eventual recovery. Protection can be afforded by social resources such as received and perceived social support and levels of social capital. Social capital describes the pattern and connections of social networks among individuals, and the shared values that arise from those networks. Following Hurricane Katrina, social capital was measured in terms of social interactions before and after the hurricane to identify predictors of health outcomes; findings support the evidence that social capital in positive forms can result in positive health outcomes (Beaudoin 2007). Smith (1996) also reports active coping as being associated with less psychological distress following flooding, while avoiding coping was associated with greater psychological distress.

This suggests that psychosocial resources should be targeted at those marginalized households who often have less access to information, support and communication channels relating to assistance programmes. Psychological First Aid has also been reported as one method effective in aiding recovery and was widely used following Hurricane Katrina (Combs 2007). This focuses on practically meeting each individual's crisis-related psycho-bio-social needs through the process of listening and formulating an action plan for recovery.

Wider community impacts

Disaster events and the recovery process can be considered as social and communal phenomena, not just affecting individuals. Floods can impair the quality of community life due to the disruption of community activities and a sense of community breakdown (Norris *et al*. 2001; Tapsell and Tunsall 2001). However, floods can also result in a positive sense of communities pulling together and helping each other, enabling mutual practical and emotional support (Tapsell and Tunstall 2001; Fernández-Bilbao *et al*. 2008; Pitt 2008). Community networks are often effective tools in aiding recovery and reducing psychosocial impacts. In Alberta, Canada, residents reported that their communities were more helpful in dealing with post-flood health problems than public and service delivery sectors (Acharya *et al*. 2007).

Restoring the social fabric of communities is therefore important in responding effectively to the psychosocial and mental health effects of disasters. Information and activities that normalize reactions, protect social and community resources, and signpost access to additional services are fundamental to effective psychosocial responses (Williams *et al*. 2009). Post-disaster response may thus be better aimed at mobilizing, maintaining and enhancing natural community and social support systems. Pelling (1997, 1998) identifies households and communities as active agents in the management of vulnerability to hazards. Involving communities in planning and implementing responses will not only give people a sense of partnership and ownership in managing emergencies and recovery but may also reduce the uncertainties and anxieties associated with flooding.

Mitigation Before and After: Adaptation

A number of factors affecting socio-psychological dimensions of flood management and risk mitigation can be identified during the period following flooding, and possibly impacting upon future preparedness.

Anxiety, cause of flood and trust in authorities and structural measures

The maintenance of high levels of anxiety following flooding may influence whether people take actions to mitigate the risk of future events. One of the hallmark symptoms of post-traumatic stress disorder is physiological reactivity to traumatic reminders, such as heavy rainfall. Studies have repeatedly cited respondents experiencing anxiety when it rains heavily and when storms are forecast; the most common behavioural response reported is the monitoring of river levels (e.g. see Tapsell *et al*. 1999; Tapsell and Tunstall 2001). These behavioural responses may be based upon logical searches for the causes of a flood and remedies for alleviating future flooding. Perception of the cause of flooding can be seen to have an impact on perception of future risk as well as on whether measures are taken to prepare for that risk.

Loss of trust and confidence in local authorities has been linked to the belief that flooding of properties was not a 'natural' occurrence but due to bad flood management, poor drainage management, and inappropriate development within floodplains rather than to climatic or weather factors. The public often differ from flood risk managers in their views on the cause of flooding and may not trust the institutions communicating or managing the risks. Thus a 'culture of blame' may develop with flood risks being open to social definition and different interpretations and constructions (Cutter 1993). Lack of trust in responsible authorities can be a significant factor affecting socio-psychological responses to floods and can impact upon how people engage with risk information they receive from these sources.

> I've got everything moved upstairs. I thought, I've got to make sure because I don't believe anything that they tell us, anyone...
>
> **Resident, Rotherham (GfK NOP 2007, p. 22)**

Personal responsibility and flood resilience measures

Although the implications of recent policy shifts from flood defence to flood risk management may be well understood by professionals, there is increasing evidence that this shift has yet to reach the ordinary citizen, who still wants (and indeed expects) to be protected, particularly when they are paying taxes to governments and local authorities. The public are generally ignorant of policies such as *Making Space for Water* in England (Defra 2005) and the new *Water Law* in Saxony, Germany (Steinführer and Kuhlicke 2007).

Evidence from Italy, Germany, the UK and Canada has shown that those living and working in at-risk areas often place faith in structural flood defences and show a distinct preference for structural mitigation measures. This can result in a false sense of security and a failure of people to act to protect themselves and their properties (Morris-Oswald and Sinclair 2005; De Marchi *et al.* 2007; Steinführer and Kuhlicke 2007; Tunstall *et al.* 2007; Werritty *et al.* 2007). Following the 2007 UK floods, people demonstrated a complacent attitude towards flooding, preferring to defer responsibility for managing flood risk to the relevant authorities (Pitt 2008).

> It's entirely the council's responsibility to prevent and deal with flooding.
>
> **Business, Hull (GfK NOP 2007, p. 21)**

Lack of awareness of the causes of floods, denial of risk, protection of emotional security, fear of reduced property prices and ignorance of appropriate mitigating actions may all be factors influencing the lack of personal responsibility for FRM demonstrated by individuals and communities. This highlights the need for better information about future flood risk and what is and is not possible. However, communication with the public needs first to reassure before it begins to inform. Anxiety management is thus seen as a barrier to public involvement in FRM. Policy-makers need to understand the importance of this for householder responses to flood risk. Harries (2008) suggests that most householders will only take action to protect themselves if they feel confident that such action will not increase their anxiety. More reliable and less stigmatizing ways need to be found by which people can increase the resilience and protection of homes without threatening their social identity. Harries (2008) suggests three approaches:

- provide tailored, independent advice to property owners;
- normalization of particular mitigation measures;
- normalization of the notions of flood risk mitigation and of proactive response.

Property-level resistance and resilience measures can be used in certain circumstances to mitigate flood damages, reduce recovery time and thus prepare people for future flooding. These measures include permanent and temporary flood-proofing measures, the function of which is to reduce the amount of ingress of floodwaters into properties. Take-up of these products and measures is generally low, partly because many people believe that it is the responsibility of government and local authorities to provide protection from flood risk (Pitt 2008). Cost is also a factor that puts products out of reach for low-income households, particularly those renting properties who also have less incentive. Insurance companies' policies on 'betterment' of properties following flooding can also restrict take-up of these measures. However, psychological variables (perceived vulnerability, risk perception and social trust) have been found to be stronger predictors for mitigation intentions than socioeconomic variables (Lin *et al.* 2008).

Local context, political commitment and governance

The importance of local contexts and governance issues that may affect decisions on FRM has also been highlighted in research from the USA (Moore *et al.* 2004) and Europe (De Marchi *et al.* 2007; Steinführer and Kuhlicke 2007; Tunstall *et al.* 2007). Government policies and decision-making processes on funding for FRM impact upon issues of inequality and social justice

(Johnson *et al.* 2007). Political commitment and perceived fairness in decision-making is therefore crucial in building local confidence in future FRM. Patterns of leadership can also be seen to impact on the management of flood risk (Morris-Oswald and Sinclair 2005). In Australia, community participation has for many years helped to enhance community preparedness and individual responsibility (Rohrmann 2003). This further raises the importance of local stakeholder and community engagement in decision-making processes (see Chapter18).

Individual and Societal Factors

A final range of factors can be identified that have an overarching impact at every stage of the hazard cycle and upon socio-psychological functioning, and which need to be considered in FRM. These factors include the individual and societal factors highlighted in Figure 20.1. It might be expected that specific social groups within communities, for example older residents, long-term ill or disabled, those on lower incomes, and minorities, will be particularly vulnerable during flood events (Fielding *et al.* 2006). These groups may (but not necessarily under all conditions) need specific targeting and support. Social vulnerability to hazards is often derived from the political, social and economic context (Blaikie *et al.* 1994; Cannon 2000; Parker 2000). Those who are most vulnerable socially, politically and economically are likely to be the least resilient in recovering from floods, and may experience the most pronounced impacts. Green *et al.* (2007) report that pre- and post-disaster inequalities slowed recovery in New Orleans following Hurricane Katrina, and suggest that structural damage was not the only, or even the primary, impediment to recovery for many residents; instead it was the outcome of pre-existing social and economic marginalization. The study provides lessons on the potential effects of recovery planning on returning residents and neighbourhoods.

Rose (1993), Fordham (1998) and others have argued that the unequal social construction of gender roles in the home may result in floods having greater impacts upon women than men. There is certainly evidence that women may experience more post-flood impacts than men, often due to their greater domestic responsibilities, which are increased or disrupted following flooding (Fordham 1998; Tapsell *et al.* 1999; Tapsell and Tunstall 2001). The evidence on age is inconclusive and effects may differ according to country and cultural contexts. Younger, working adults with young children may often suffer increased stress levels as they have more responsibilities. Some studies have found older people to be more psychologically resilient to flooding, and there are indications that ethnic minority groups may be more adversely impacted due to language difficulties, low incomes and ignorance of political processes (Tapsell *et al.* 1999). Pre-disaster functioning and individual personality may also influence outcomes. Otto et al. (2006) found that in a study of the 2002 floods in Dresden, Germany, a positive outlook on life and people's belief in a just world were able to buffer psychological symptoms following a natural disaster. There is also evidence for certain groups being more vulnerable at certain **phases** of a flood event than others (De Marchi *et al.* 2007, pp. 188–90; Steinführer and Kuhlicke 2007, pp. 113–5; Tunstall *et al.* 2007, pp. 125–7). Different factors come into play in the various phases of an event and affect specific behavioural responses and coping activities. Local context is a key factor: both local conditions and event specifics.

Finally, it is suggested that values (norms and beliefs) can be deeply relevant to understanding community level response to flood hazard (Morris-Oswald and Sinclair 2005). Awareness campaigns that reflect social values and perspectives are likely to be the most effective (Shaw *et al.* 2005; Renn 2008). Shared values indicate common motivations and can serve as the common ground to achieve common goals, for example increased resilience. The Environment Agency in England and Wales has looked at 'values modes' approaches to inform how it could better target flood warnings to people in different flood risk situations, for example through better messaging and calls for behaviour change. Through large-scale surveys populations can be segmented into different groups based upon people's values and attitudes in an attempt to measure why people do what they do. This provides a way of understanding how to change or reinforce expectations, attitudes or behaviours (Campaign Strategy Ltd. 2007). Understanding communities and their values is therefore crucial to future FRM and improving stakeholder engagement. This raises non-technical communication challenges that have nothing to do with water or floods but are about **people**.

Conclusions: New Directions for Flood Risk Management?

As the above discussion highlights, floods and decisions around FRM have the potential to seriously impact upon the social, psychological and physical well-being of those affected and impact upon their social and psychological functioning. The framework utilized in this chapter provides a way of identifying and understanding the socio-psychological dimensions of FRM at each stage of the disaster cycle. From the above discussion a number of recommendations can be drawn to inform FRM policymakers and practitioners, including local authorities. Table 20.1 draws out the key points from the above discussion in relation to the phases of the disaster cycle, and puts forward a set of practical recommendations organized around the key themes identified. Some of these recommendations overlap between both issues and the phases of the disaster cycle. In the UK many of these suggestions are currently being explored or implemented, particularly since the summer 2007 floods and subsequent Pitt Review, while others require further investigation and action. However, FRM practitioners and policy-makers in other countries may be able to draw upon these suggestions and assess their relevance in relation to their own particular circumstances and local contexts.

A key question for policy-makers is how risk attitudes and risk perceptions are formed, and which means of communication impact upon and shape such attitudes and perceptions. The recruitment of experienced social scientists, including sociologists and psychologists, in helping to better understand how people construct risk, and their motivations and behavioural responses to such risk, is essential. Emotional considerations and how they influence behavioural responses to flood hazard comprise another important area for future research. The emotional and psychological impacts of flooding are often identified as the major barriers to recovery, although this is also one of the areas where past support and research has been most lacking. Also important is the recognition of the long-term mental health impacts and the need for better information on those liable to suffer such impacts in order that they can be located and targeted for assistance.

Risks also need to be viewed in the context of evaluations of local life and the local environment. Experiences from Europe, Australia and the USA indicate that 'bottom-up' flood incident strategies are needed that are designed around detailed understanding of the socioeconomic and institutional characteristics of each area. Interventions are more likely to be successful when the emphasis is upon building local knowledge and augmenting existing capacity. Effective FRM will only be successful with the involvement of the public and relevant stakeholders. Risk reduction measures need to be tailored to the highly differentiated risk and risk awareness levels between and within countries; a 'one size fits all' approach will not work. Stronger engagement of citizens in risk management efforts should contribute to raising risk awareness and disaster preparedness and can enhance the acceptance of, and responsibility for, risk reduction measures. Genuine community involvement is likely to produce valuable and tangible outcomes and long-term benefits; it promotes understanding and ownership and enhances commitment, with people learning by doing rather than through the receipt of passive information. Funding agencies therefore also need to channel resilience-building support and vulnerability reduction efforts into education, capacity building, psychosocial programmes and people-centred strategies.

Table 20.1 Key issues and recommendations on the socio-psychological dimensions of flood risk management (FRM) in relation to the phases of the disaster cycle

Preparedness before event (links with mitigation after event)	
RISK AWARENESS AND CONSTRUCTION *Issues:* Low levels of awareness and acceptance of risk The need to engage people and encourage them to prepare and plan for floods	*Recommendations:* • Build upon existing community flood experience and local flood histories; where none exist, highlight impacts and experience from other similar locations and communities • Provide information on causes of different types of floods and what actions can be taken to prepare • Better explain the probabilities of flooding. Use simple terminology and avoid terms like 1 : 100 • Emphasize the consequences of floods not just the probabilities • Include information on local flood history to potential purchasers of properties • Target landlords, letting agents and renters in awareness-raising campaigns • Identify and target existing social networks to help increase awareness and community preparedness • Encourage the development of household, community and business plans for flooding • Encourage and support more community flood warden schemes • Draw upon the expertise of psychologists to increase understanding of the role of risk construction, attitudes and perceptions in individual decision-making behaviour and motivations, and in formulating more effective response strategies • Consider the further exploration of 'values modes' approaches to inform communication strategies
THE ROLE OF EMOTIONS *Issues:* Need for better acknowledgement, recognition and understanding of the role of emotions in people's risk construction Need to understand people's perception of home/place and how flooding undermines such perceptions and identities	*Recommendations:* • Draw upon the expertise of psychologists to: increase understanding of the role of emotions in risk construction and their impact on risk preparedness and response; and to help understand people's needs to protect their emotional security and representations of security • Build upon people's strong emotional attachment to the home in awareness raising and preparedness campaigns • Develop strategies to normalize the threat of flooding • Develop more innovative property resistance measures that have less visual impact and are not perceived to reduce the value of properties
INSURANCE *Issue:* Low take-up of insurance among certain groups in `at-risk' areas	*Recommendations:* • To facilitate insurance uptake among low-income groups and renters, need to emphasize the benefits of insurance in awareness raising by highlighting the costs of repairing homes and replacing possessions • Encourage institutional pressures to increase insurance take-up
Response and relief during event	
FLOOD WARNINGS *Issues:* Low take-up of flood warning services Inappropriate response to warnings	*Recommendations:* • Raise awareness of the benefits of flood warnings in reducing potential losses and distress • Tailor flood warning methods according to the type of flood risk and to recipient needs and location characteristics • Explore the feasibilities for more face-to-face warnings, e.g. from flood wardens and door knocking • Do not overly rely on all people having access to the internet as a medium for warning and information provision • Take a `response focus' to warnings to facilitating effective actions
RISK TO LIFE *Issue:* *Mortality in floods is often associated with risk-taking behaviour, especially among younger males*	*Recommendations:* • Using graphic images and examples emphasize the power of flood waters and the various dangers of walking or driving through floodwaters • In driving test examinations include questions on the dangers of floodwaters, e.g. at which of the following depths would a large four-wheel drive vehicle be carried away by floodwaters?

(*continued*)

Table 20.1 (*Continued*)

Response and relief during event	
EVACUATION *Issue:* Poorly organized evacuation can increase the stress experienced by those affected by flooding	*Recommendations:* • Better coordinated and organized emergency response and evacuation plans • Re-evaluate the location of evacuation centres and emergency response control centres outside risk areas • Do not assume that everyone has access to transportation for evacuation • Consider cultural sensitivities in evacuation centres in areas with large ethnic minority groups, e.g. segregated areas and facilities for males and females, provision of appropriate food
Recovery after event	
POST-FLOOD DISRUPTION AND FINANCIAL WORRIES *Issue:* Psychological distress is often more reflective of the difficulties and hardships encountered during the recovery rather than the impact phase of an event	*Recommendations:* • Provide practical support to help during clean-up • Ensure contingency planning for the loss of essential services and provision of essential supplies (e.g. food and water) in major emergencies • Make available definitive guidance on best practice for drying, stripping out and repairing of properties • Provide more advice and support for small businesses • Encourage employers (including local authorities) to be more sympathetic to staff who need to take time off to deal with recovery • Work with insurance companies, loss adjusters and building contractors to improve their assessment and handling of insurance claims and the repair of damaged properties, and their sensitivity to those who have been flooded
HEALTH *Issues:* Need clearer understanding of the effects of flooding upon people's physical and mental health and well-being and how to mitigate these	*Recommendations:* • Further research into the risk to public health from floods • Explore the feasibility of increased use of Psychological First Aid to practically meet people's individual psycho-bio-social needs through the process of listening and formulating an action plan for recovery • Facilitate post-event self-help support groups where people can meet and talk with others who have been affected • Explore ways to facilitate the building of social networks and resources • Encourage active coping to reduce psychological distress following flooding • Draw upon those who have experienced past flood events to demonstrate that people can and do recover • Target psychosocial resources at marginalized households with less access to information, support and communication channels relating to assistance programmes • Explore ways to highlight the importance of keeping up routine social activities to help maintain social networks and support camaraderie during recovery • Increase understanding of which groups may be more vulnerable at certain phases of a flood event than others • Conduct research to better understand the impacts of floods on front-line workers
WIDER COMMUNITY IMPACTS *Issue:* Disaster events and the recovery process need to be recognized and considered as social and communal phenomena and not just affecting individuals	*Recommendations:* • Assess the wider recovery needs of affected communities; practical, social and emotional • Better understand communities and their values, as awareness campaigns that reflect social values and perspectives and common motivations can serve to achieve common goals, e.g. increased resilience • Encourage a positive sense of community through facilitating community networks • Prioritize restoring the social fabric of communities (e.g. by setting up meeting places) to mobilize, maintain and enhance natural community and social support systems • Involve communities in planning and implementing responses

Table 20.1 (*Continued*)

Mitigation after event and before next event	
ANXIETY REDUCTION, CAUSE OF FLOOD AND TRUST IN AUTHORITIES *Issue:* Anxiety following flooding can be a barrier to public involvement in FRM and encouraging preparedness planning and actions	*Recommendations:* • Communication with the public needs first to reassure before it begins to inform • Policymakers need to understand the importance of anxiety management in householder responses to flood risk • Work with communities to foster increased levels of trust and to reduce anxieties surrounding flooding • Explore more effective and innovative ways to provide information on the causes of flooding and future flood risk and what is and is not possible for preventing or mitigating such risk
PERSONAL RESPONSIBILITY AND FLOOD RESISTANCE AND RESILIENCE MEASURES *Issues:* Denial of risk, protection of emotional security, fear of reduced property prices and ignorance of appropriate mitigating actions may all be factors influencing the lack of personal responsibility Faith in structural flood defences and preference for structural mitigation measures Deference of responsibility to relevant authorities	*Recommendations:* • Draw upon the expertise of psychologists to better understand the psychological factors (perceived vulnerability, risk perception and social trust) found to be strong predictors for mitigation intentions • Put more emphasis on helping people to understand the causes of floods and what is and is not possible for preventing or mitigating the risks • Explore more reliable and less stigmatizing ways by which people can increase the resilience and protection of homes without increasing their anxiety and threatening their social identity • Encourage normalization of particular mitigation measures to increase take-up of such measures • Facilitate the normalization of the notions of flood risk mitigation and of proactive response through awareness-raising activities • Explore the provision of tailored, independent advice for property owners • Encourage insurance companies to review their `like for like` policies and include provision for `betterment' of properties • Facilitate increased stakeholder and community engagement in local decision-making processes

There is also a need to increase public recognition that it is not possible to protect from all floods and to engage people to take some responsibility (within their means) for their own protection. Public awareness of the potential **consequences** of flooding, including the length of the recovery process, and the residual risk of flooding even where defences are in place, is also necessary. Importantly, there is a need to normalize the threat of floods and to build future resilience, rather than just fixing what a flood has damaged and returning to a pre-flood state. This is now becoming increasingly recognized in the UK and elsewhere. However, in policy terms there are tensions and contradictions associated with 'improving resilience'. The overall philosophy of flood management in many locations is still one of a technical fix with resistance and recovery geared towards preserving and reinstating 'normality'. However, 'putting things back to normal' may simply reproduce existing vulnerabilities to flooding that are embedded in social structures and practices, hence the need to develop a new resilience and normality. Although pre-existing social and economic vulnerabilities and inequalities that intensify flood loss and disruption need to be tackled, these are normally outside the control of flood risk managers. However, allocations of funding, social justice and fairness in FRM decision-making is within the remit of such managers and policymakers.

A rebalancing of flood management policy is needed with more emphasis given to resilience-building adaptation, while recognizing that more traditional strategies designed to resist flooding and provide emergency relief will continue to be needed over the short and medium term. Twigger-Ross *et al*. (2008) call for adoption of a socio-technological approach to FRM. Moreover, non-technical approaches need to be better reflected in the allocation of FRM resources; socio-psychological means and resources also need to be considered. If future FRM and mitigation is to be socially sustainable there is a need to assess the cost-benefit of socio-psychological measures versus technical measures, as the former may often be cheaper and provide longer-lasting benefits for community safety and coping with flood impacts. Comprehensive preparedness of residents and institutions before a flood event, and optimum behaviour and survival during and after an event, cannot be achieved without careful socio-psychological grounding.

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21 Assessment of Infection Risks due to Urban Flooding

LORNA FEWTRELL, KEREN SMITH
AND DAVID KAY

Introduction

Urban flooding can result in a number of public health impacts, including fatalities resulting from drowning and entrapment during the immediate flood event, mild impacts such as stomach upsets, and long-term effects such as increased levels of anxiety and mental health problems (Fewtrell and Kay 2008). This chapter focuses on the risk of human infection presented by microbial hazards in the flood water.

For there to be an infection risk from urban flooding, a number of factors must coincide; these can be thought of in terms of a Source, Pathway and Receptor paradigm, as illustrated in Figure 21.1.

Thus, the Source is the microbially polluted flood water, the Pathway is the route that the flood water takes, and the means by which people are exposed to its microbial contaminants, and the Receptor is the householder (or member of the rescue services, etc.) affected by the floods and exposed to the constituent pathogens. The source, pathways and receptor issues are examined in more detail, before the presentation of an exploratory quantitative microbial risk assessment (QMRA), which examines the risk of gastrointestinal disease resulting from flooding and associated clean-up operations. The QMRA approach has been adopted to gain a 'feel' for the infection risk as generally data are not collected to allow specific estimation via other means.

Source

In an urban flood there may be a number of component flood flows that contribute to the 'quality' of the flood water. These are illustrated in Figure 21.2 and outlined in more detail in the following subsection.

Component flood flows

Combined sewer overflows

Combined sewer overflows (CSOs) have been identified as an important component of microbial discharge from urban areas, representing a mixture of raw sewage, stormwater microbes and combined sewer sediment. Ellis and Yu (1995) identified CSOs as a primary source of microbial contamination to urban waters in London.

Foul flow

The occurrence and concentration of microbial contamination in raw sewage will vary, both temporally and spatially, depending upon the catchment and the health and size of the population. Treated, or partially treated, wastewater may also be a component of a flood flow after discharges from wastewater treatment works into receiving

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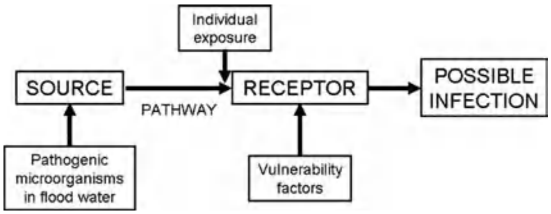

Fig. 21.1 Source, pathway, receptor model.

waters, or where heavy rainfall results in overloading of the treatment works (Kay *et al.* 2008a, 2008b).

Rivers

Rivers receive microbial pollution from a number of sources, including most of the key flooding components identified in Figure 21.2. The relative contributions that are received from each component vary according to antecedent rainfall patterns, season and individual catchment characteristics.

Rural diffuse

In urban areas, rural diffuse pollution will be largely represented as an upriver boundary condition. Microbial pollution from rural catchment sources is likely to predominate in upstream areas and be transported downstream to larger urban areas by the fluvial network. Rural pollution from agricultural upstream areas encompasses diffuse overland flow (Doran and Linn 1979; Kay *et al.* 2008b) and subsurface flow (Rahe *et al.* 1978; Unc and Goss 2003) to watercourses as well as direct deposition to surface waters where livestock have access to them (Collins 2004; Kay *et al.* 2007a, 2007b). Diffuse microbial sources associated with runoff from pasture land are likely to be supplemented in the lowlands by applications of slurry, farmyard manure and sewage sludge to arable land, as well as contamination from point sources such as runoff from farmyards and animal waste storage facilities on livestock farms (Geldreich 1996; Jones and Obiri-Danso 1999; Obiri-Danso and Jones 1999; Crowther *et al.* 2002; Collins 2004; Edwards and Kay 2008; Stapleton *et al.* 2008).

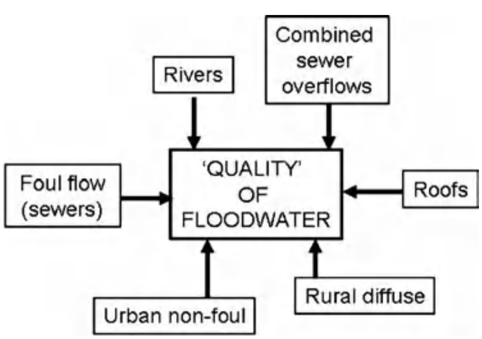

Fig. 21.2 Component flood flows.

Urban 'non-foul' (including roof discharge)

Urban non-point-source pollution has been increasingly recognized for its impact on the microbial quality of receiving water, especially under high or storm flows. Urban diffuse sources of microbial pollution to the surface drainage system include surface runoff from impervious surfaces (including roofs), misconnections of foul sewers into surface water drains and additionally, in the UK, dual manholes allowing foul sewer blockages to cause overspills into the surface water drainage system (O'Keefe *et al.* 2005; Fewtrell and Kay 2007; Stapleton *et al.* 2008).

Microbial 'quality'

A review of microbial concentrations in flood components was conducted by searching a number of web-based databases including PubMed and ISI Web of Knowledge, search engines including Google and Google Scholar, and individual relevant websites, including the UK's Department for Environment, Food and Rural Affairs (Defra), the Environment Agency and the World Health Organization. The main focus of the review was pathogen concentrations, but faecal indicator concentrations were also examined (Smith *et al.* 2007).

Faecal indicator organisms

Microbial quality is usually assessed by measuring 'faecal indicator bacteria' (also referred to as faecal indicator organisms, or FIOs). Thermotolerant coliforms (also termed faecal coliforms), *Escherichia coli* and intestinal enterococci (previously termed faecal streptococci) are generally harmless bacteria (although *E. coli* O157 is pathogenic) that are present in high numbers in faecal material and are the most commonly examined FIOs. Thus, their presence in water samples is used to indicate the presence of faecal pollution and the possibility that faecally associated pathogens may also be present (see below). Other, less frequently utilized, FIOs include *Clostridium perfringens* and coliphage organisms. The ecology and environmental survival characteristics of bacterial, viral and protozoan pathogens (see below) vary and so there is no single ideal indicator organism (Savichtcheva and Okabe 2006).

Pathogens

Pathogens are infectious microorganisms that can cause infection and disease. They can be classified as bacterial (e.g. *Salmonella*), protozoal (e.g. *Cryptosporidium*), viral (e.g. enteroviruses), fungal (e.g. *Candida*) or helminth (e.g. *Trichobilharzia*). Pathogens may be introduced to the water as a result of contamination (often from faecal material) or may be naturally present in the environment (i.e. autochthonous). The pathogens identified in the literature review are outlined by flood component in Table 21.1. Pathogens present in the water as a result of contamination generally do not grow in the environment, and thus concentrations decrease with time. Those naturally present in the environment can increase in numbers in flood waters. The pathogens identified in Table 21.1 reflect, to a large degree, those microorganisms for which routine tests are available. Thus, the absence of a 'yes' for any entry may, in many cases, mean that the pathogen has not been

Table 21.1 Pathogens by flood component

		Flood component						
	Pathogen	Rivers	Sewage	CSO	Urban non-foul	Roofs	Rural diffuse	UK literature
Bacteria	*Aeromonas*	yes				yes		
	Campylobacter	yes	yes			yes	yes	yes
	Escherichia coli O157	yes						
	Helicobacter pylori	yes	yes					
	Legionella					yes		
	Listeria	yes	yes					yes
	MAC	yes				yes		yes
	Pseudomonas	yes	yes		yes	yes		yes
	Salmonella	yes	yes	yes	yes	yes		yes
	Staphylococcus				yes			
	Yersinia	yes					yes	
Protozoa	*Cryptosporidium*	yes	yes	yes		yes	yes	yes
	Giardia	yes	yes	yes		yes	yes	yes
Viruses	Adenoviruses	yes	yes					
	Astrovirus	yes	yes					
	Enterovirus	yes	yes		yes			
	Hepatitis A	yes	yes					
	Norovirus	yes	yes					
	Rotavirus	yes	yes					

CSO, combined sewer overflows; MAC, *Mycobacterium avium* complex.

looked for in that component. Notable microorganisms missing from Table 21.1 include *Leptospira* spp. and *Shigella* spp., both of which are likely to be found in some flood components in the UK.

Pathways or Routes of Exposure

A person may be exposed to microorganisms in flood waters and sediments through various routes during the flood itself, throughout any clean-up procedures and also, in some cases, after the remediation period when flooded residences are reinhabited. Exposure may be via three main pathways: ingestion, inhalation or skin (or wound) infection.

• Skin or wound infection is most likely to occur either during the flood (e.g. during evacuation), or in the immediate period of remediation when people would potentially be directly exposed to flood waters and, thus, experience contact with water. Reports of flood-related infection via this route are uncommon, although in the USA a fatal case of septicaemia (blood poisoning) in a 57-year-old man was attributed to infection of an open wound and contact with flood water (Spice 2004). The victim was also diabetic, which would have made him more susceptible to infection.

• Inhalation of microorganisms may occur during flooding and the remediation period, via flood water sprays and airborne water droplets. It is also possible that some pathogens, such as noroviruses and *Mycobacterium avium* complex (MAC), may remain in house dusts after the flood waters have dried, although there is little evidence to suggest significant survival of these pathogens. Inhalation of mould and fungal growth, in particular, may have implications for respiratory health. Immunocompromised people may be at risk of systemic infection, and exposure to home moulds is thought to be a factor leading to cases of bronchial asthma, chronic allergic rhinitis, hypersensitivity pneumonitis and sick building syndrome (Kramer *et al.* 2000). However, although the growth of mould and fungi may be a consequence of flood inundation, it is not related to the pathogen content of flood waters, and is not considered further here.

• Reports of aerosol flood-related infection are rare, although a small outbreak of Legionnaire's disease has been associated with a flood incident (Kool *et al.* 1998). In the USA, the owner, a waitress and a regular customer of a bar became ill with Legionnaire's disease; investigations revealed that the only common exposure for the cases was the bar, which had been flooded nine days before the first reported illness. *Legionella pneumophila*, of the same serotype isolated from the cases, was isolated from the sump in the crawlspace under the bar and from a hose running from the sump to the street. Following the flooding, water was pumped from beneath the bar to the street over a period of three days by an old electrical sump pump. The pump generated a considerable amount of heat while running and it is thought that this may have provided temperatures favourable to the growth of *Legionella*. This, in conjunction with an aerosol generated from a jet of water coming from a small hole in the side of the pump, was probably responsible for the outbreak.

• In terms of likely infection in any flooded population, the most probable route of exposure to flood-associated pathogens is by accidental ingestion. There is epidemiological evidence of infection resulting from direct contact with flood waters. For example, an outbreak of norovirus infection (a relatively mild, self-limiting stomach upset) in American tourists in Salzburg, Austria, revealed a link with direct exposure to flood water contaminated with raw sewage (Schmid *et al.* 2005). Following heavy exposure to the flood waters during the clean-up, 49 of 64 people in the party (77%) succumbed to a gastrointestinal illness later diagnosed as norovirus infection. The infections were thought to be contracted through direct contact with the flood water, followed by secondary person-to-person transmission.

• After the cessation of flooding and the completion of the clean-up process, ingestion of soil may be a potentially important route of exposure to flood-deposited pathogens, via recreational activities, gardening and the consumption of home-grown foods (Davis and Mirick 2006). Young children may be at particular risk of pathogen ingestion from soils due to their mouthing behaviour and frequent hand-to-mouth contact.

• The ingestion of contaminated water supplies, particularly where private potable supplies are used in inundated areas, may be a significant exposure risk (Petrie *et al.* 1994; Duke *et al.* 1996; Fewtrell *et al.* 1998; DWI 2003; Kay *et al.* 2007c). However, in the UK, the majority of homes in the urban environment receive mains water from a water utility company, where the drinking water supply will often be a source outside the urban area and unaffected by flooding. Where the infrastructure is adequate, outbreaks of illness are unlikely to result from flooding unless the security of the water supply is directly compromised (Hunter 2003). Where public drinking water supplies are compromised, the water utility is required to provide alternative sources of water, as illustrated in the floods in Gloucestershire in summer 2007 when the Mythe water treatment works in Tewkesbury was flooded leaving 350,000 people to be supplied by bowser (Fig. 21.3) and bottled water. The alternative source is, however, rarely adequate for all uses, possibly leading to compromised personal hygiene.

Receptors

Receptors are the people subject to the flood event or exposed to the flood water. Clearly, physical exposure is the first step in defining the receptor population. However, this may not reflect the susceptibility of different populations groups to infection. Key population segments of interest are the very young and old as well as those on drugs that depress the immune system (such as those given to transplant patients) and those with HIV infection.

Vulnerable groups

There are a number of factors that make some people more vulnerable to health impacts than others. These may be related to age, pre-existing disease, and behaviour, as illustrated in Table 21.2.

Quantitative Microbial Risk Assessment (QMRA)

Data on a number of infections are routinely collected; however, ascribing illness to an actual flooding event is difficult, thus QMRA can be used to provide an estimate of flood-related infection risk. QMRA is a formal probabilistic process for estimating microbial risks within defined scenarios. There are four steps (in which the first three are combined in order to characterize the estimated risk), namely: hazard characterization, dose-response assessment, exposure assessment, and ultimately risk characterization.

Fig. 21.3 Bowser supply in Gloucester, following flooding of Mythe water treatment works. Photo: L. Fewtrell.

Table 21.2 Selected factors leading to inequality of health risk

Factor	Effects
Age	The very young and old are more likely to acquire infections due to naive or waning immunity and, once infected, are more likely to develop more severe outcomes
Pre-existing disease	A person with AIDS, severe combined immunodeficiency syndrome, diabetes, heart disease, cancer, etc. is more likely to be vulnerable to infection and more likely to suffer severe symptoms
Genetic	People with certain genotypes are more likely to experience complications, such as joint problems, following gastrointestinal illness
Gender/pregnancy	Certain infections are more severe in pregnancy, either increasing the risk of fatality for the mother or resulting in damage to the fetus
Behaviour	Behaviour [such as a refusal to evacuate or geophagy (soil eating)] may result in higher exposure to infectious agents

Methods

Where possible, parameters for inclusion in the QMRA were represented as probability distributions (see following sections) rather than point estimates, in order to examine the effects of uncertainty. Monte Carlo sampling (5000 iterations) was used for each simulation run using @Risk version 4.5 Professional edition software (Palisade Corporation 2002). The resulting estimates of infection were quantified in terms of disability-adjusted life years (DALYs) in order to give an overall health impact and allow comparisons between different scenarios. DALYs are summary measures of health that allow the comparison of effects across a wide range of health outcomes, including mortality and morbidity. The measure combines years of life lost by premature mortality (YLL), with years (or days, weeks or months) lived with a disability (YLD), standardized using severity or disability weights. The weights range from 0 (perfect health) to 1 (dead). The calculation of YLL due to premature mortality requires an estimate of life expectancy. This varies according to age group and gender, but in the UK overall average life expectancy at birth is 79 (GAD 2007).

Flood scenario

The demographic profile of the flooded population will obviously be case-specific and may be accounted for when such variables are known. For the purposes of illustration, a hypothetical scenario was based on the assumption that 400 houses were flooded (mean household composition of 2.57 people) with the population characteristics shown in Table 21.3 (based on an urban population in the north of England, using data from the 2001 census).

Based on the age distribution and the mid-point of the age groups, the average age of the population is 35.6. A total of 20.25% of the population have a limiting long-term illness.

In this example, it was assumed that this population was flooded with a mixture of river water (80%), raw sewage (10%) and urban runoff (10%). The scenario investigated was the risk of illness associated with the flood clean-up process.

Pathogen choice

There are far too many pathogens, and a lack of data about most, to consider all possible causes of gastrointestinal infection that could be present in flood water. A common approach is, therefore, to consider a number of reference pathogens (WHO 2004), usually consisting of a bacterial, viral and protozoan pathogen. Suitable reference

Table 21.3 Population characteristics for an urban flood affecting 400 households

	People	
Age (years)	%	Number
0–4	7.68	80
5–9	7.93	80
10–14	8.03	84
15–65	63.85	656
66–74	6.35	65
75 +	6.16	63
Total	100	1028

pathogens are usually those that present a worst case combination (NRMMC 2006) of high occurrence, relatively high concentrations in flood components and high pathogenicity.

Based on an examination of the literature relating to flood water quality (Smith *et al.* 2007, Table 1) the most frequently identified bacterial and protozoan pathogens are *Campylobacter* spp., *Salmonella* spp., *Cryptosporidium* spp. and *Giardia* spp. Campylobacters are the most commonly reported cause of bacterial gastroenteritis in the UK, with an estimated incidence of 8.7/1000 population (Adak *et al.* 2002) and, on this basis, were chosen as the bacterial representative. *Cryptosporidium* infection is more prevalent than *Giardia* infection (estimated incidence in the UK of 8.1/10,000 and 5.4/10,000 respectively: Adak *et al.* 2002). *Cryptosporidium* has also been more frequently associated with waterborne illness than *Giardia* in the UK, and was chosen as the protozoan representative.

The choice for virus was less clear cut. Noroviruses are the most common cause of viral gastroenteritis in the UK, with an estimated annual incidence rate of 12.1/1000 (Adak *et al.* 2002) but there is no dose-response relationship for this virus. Rotaviruses are the second most common cause of viral gastroenteritis and a dose-response relationship has been established (Haas and Eisenberg 2001), but concentrations cannot be determined by cell culture. A composite virus was chosen based on concentrations of adenovirus (which can be determined by cell culture) with the dose-response characteristics of rotavirus.

Hazard characterization

The reference pathogens have been outlined above; this section summarizes the key characteristics of the pathogens in terms of QMRA and DALY quantification.

Campylobacter spp.

Infection with *Campylobacter* spp. proceeds to clinical illness in 30% of cases (WHO 2004). The severity weight (0.086) and duration (6 days) for uncomplicated cases of campylobacteriosis are based on values described by Havelaar *et al.* (2000a) for illness in the general population and illness reported to general practitioners. The severity weight (0.28) and duration (365 days) for complicated campylobacteriosis is based on the mean value for Guillain–Barré syndrome – a disease of the peripheral nerves and occasional complication of campylobacter infection (adapted from Havelaar *et al.* 2000a), with the incidence based on the campylobacteriosis hospitalization rate of 0.5% reported by Mead *et al.* (1999). Recent mortality statistics (2001–2005) from the UK indicate that fatalities from campylobacteriosis occur in people aged over 65. An age-related case-fatality rate of 0.0083% for the UK was estimated, with a mean age of death of 80 (ONS 2007) and a resultant loss of 8.3 years of life (based on additional life expectancies for males and females aged 80 of 7.57 years and 9.03 years respectively (GAD 2007).

Cryptosporidium spp.

Infection with *Cryptosporidium* spp. in developed countries is believed to result in illness in the immunocompetent population in 71% of cases, while infection in the immunocompromised population is thought to lead to illness in virtually 100% of cases (Havelaar *et al.* 2000b). The severity weights and illness duration are 0.054 and 6 days, respectively, for illness in the normal population, and 0.13 and 47 days, respectively, in the immunocompromised population. A case-fatality rate of 0.0158% has been estimated from UK data on age-related incidence, known under-reporting of non-fatal cases and population statistics (FSA 2000; Adak *et al.* 2002; HPA 2007a; ONS 2007). The mean age at death from cryptosporidiosis is 63.7 years (this figure was assumed to be 65).

The effect of HIV/AIDS on mortality from cryptosporidiosis infection is unclear. A study reported in 1987 showed that cryptosporidiosis in patients with HIV/AIDS had a case-fatality rate that was significantly higher ($p < 0.01$) than the case-fatality rate for patients without reported HIV/AIDS (Navin and Hardy 1987). Furthermore, Ruisin *et al.* (2000) suggested that about 10–15% of AIDS patients die of complications related to cryptosporidiosis. However, examination of UK mortality data (ONS 2007) indicates a mean age of death from cryptosporidiosis of 63.7 years, and of death from HIV disease of 42 years. Over the period 2001 to 2005, in the UK, there were limited (6) deaths from cryptosporidiosis and only one incident of mortality under 50 years of age. With such limited data, it is inappropriate to try to quantify mortality from cryptosporidiosis in AIDS cases separately from that of the rest of the population. In this QMRA, the influence of HIV/AIDS was restricted to an estimation of the additional severity and duration of the infection in the immunocompromised population.

Viruses

Infection is assumed to proceed to clinical illness in 50% of cases (WHO 2004). The severity weight of 0.093 is based on an uncomplicated case of diarrhoea (VGDHS 1999). The duration of symptoms is usually between 3 and 8 days (Ruisin *et al*. 2000; HPA 2007b). Rotavirus infection is not usually fatal in people over the age of 4 years (ONS 2007), thus fatalities were only estimated for children under the age of 5 years.

Dose-Response assessment

For microbial hazards the dose-response characterizes the relationship between exposure and the incidence of the health effects, as exposure to the hazard does not necessarily mean that a health impact is inevitable. In many cases, depending on what dose is received (and how), the body may be able to remove the pathogen without any obvious ill effect. Dose-response relationships are typically characterized by exposing a population of volunteers (usually young healthy people) to various concentrations of the microorganism under investigation. The results are then modelled (generally using exponential and beta-Poisson models – for further information see Fewtrell *et al*. 2008b) to enable extrapolation to low doses. The dose-response parameters shown in Table 21.4 were used in the QMRA.

Table 21.4 Dose-response relationships

Pathogen	Beta-Poisson		Exponential	Reference
	α	β	r	
Campylobacter spp.	0.145	7.584		Medema *et al.* 1996
Cryptosporidium parvum			0.004005	Teunis *et al.* 1996
Rotavirus	0.265	0.442		Haas *et al.* 1993

Exposure assessment

The demographic profile of the flooded population was outlined above (see 'Flood scenario'). It was necessary to make a number of assumptions about exposure during the clean-up process in order to conduct the QMRA.

Clean-Up

In this example it was assumed that the majority of the residents would wait until the flood water had dropped considerably, before returning to the flooded property to begin the remediation process, and would continue to live elsewhere until the property was restored. The duration of the clean-up process will depend upon the size of the property, the number of people involved and the extent of the flood damage. The initial clean-up process (when people may be exposed to pathogens) was assumed to last between 1 and 4 days. Daily flood water contact, during this period, was assumed to be the length of the working day (up to 14 hours, with a mean daily exposure of 7 hours), with those involved returning to alternative accommodation at the end of the day. It was assumed that, where possible, children would be kept away from the clean-up, with children under the age of 5 not present during the cleaning process. A normal distribution was assumed to account for the proportion of older children present, with an average of 10% presence for children aged 5 to 9 and 20% for children aged 10 to 14. It was assumed that all household residents (aged 15 and over) would be involved in the clean-up, but no account was taken of non-resident friends and family who may have been present in a supportive capacity.

Flood water ingestion

The volume of water ingested accidentally is difficult to quantify. Westrell *et al.* (2004) assumed an ingestion of 1 mL in children playing near a source of reused wastewater and transferring water from hand-to-mouth. Tanaka *et al.* (1998) assumed that a golfer exposed to a course irrigated in wastewater will ingest 1 mL in a single exposure. It has been assumed that in a clean-up situation 1 mL per hour would be ingested in both children and adults.

Flood water 'quality'

As outlined above under 'Flood scenario', it has been assumed that the flood water consists of 80% river water, 10% urban surface runoff and 10% foul flow. The pathogen concentration ranges for the riverine and foul flow components, shown in Table 21.5, are based on a combination of literature and experimental data (Fewtrell *et al.* 2008a). The urban non-foul component was assumed to act as a diluting component, containing negligible levels of pathogens in comparison to the other components.

Risk characterization

This section brings together the estimates of exposure and dose-response for each of the identified hazards to provide an overall estimation of the risk of illness, as shown (for the mean estimates) in Table 21.6.

Table 21.5 Pathogen concentrations in the principal flood components

Pathogen	Concentration (/litre)	
	River	Foul flow
Campylobacter	0–3300	10–180,000
Cryptosporidium	0–59	1–96
Virus	1–64[a]	70–3200

[a]Based on a 50th of the sewage value.

Table 21.6 Risk characterization summary

Hazard	Exposure	Estimate	Cases of illness	DALYs
Campylobacteriosis	Clean-up	Min.	0.076	2.2×10^{-4}
		Mean	7.529	2.1×10^{-2}
		Max.	56.67	1.6×10^{-1}
Cryptosporidiosis	Clean-up	Min.	0.008	1.1×10^{-5}
		Mean	0.387	5.3×10^{-4}
		Max.	3.002	4.1×10^{-3}
Viral enteritis	Clean-up	Min.	0.141	1.8×10^{-4}
		Mean	38.16	4.8×10^{-2}
		Max.	139.9	1.7×10^{-1}
Overall illness		Mean	46	6.9×10^{-2}

It can be seen from this Table that the risk is dominated by viral enteritis, which accounted for 38 of the estimated 46 cases of illness.

Discussion

Over 20 possible pathogens have been identified that could present an infection hazard in either flood water or sediment, depending upon the composition of the flood. The majority of these pathogens cause gastrointestinal upsets. Of the non-gastrointestinal pathogens, many either cause clinical infections rarely (e.g. *Leptospira* spp.), are opportunistic pathogens (e.g. *Pseudomonas* spp. and *Staphylococcus* spp.) or have other, more common, means of transmission (e.g. *Legionella* spp. and *Listeria* spp.).

There is relatively little published information on actual behaviour during flooding and clean-up, although it is safe to say that in almost all cases some exposure is inevitable. The most frequent routes of exposure would be expected to be through ingestion and skin exposure. While skin exposure to flood waters is relatively obvious, ingestion may be less so, and is likely to occur mainly through hand-to-mouth transfer where complete immersion (e.g. during evacuation) does not occur.

The exploratory QMRA of flood clean-up-related infection suggests a relatively high number of cases of gastrointestinal illness (46 cases from a population of 1028), although these cases would generally be relatively mild and short-lived (as the bulk of cases are attributable to viral infection). The combined DALY score of 0.069 can be placed in context by considering the annual burden of illness resulting from campylobacteriosis in England and Wales from all causes. Analysis based on an estimated annual incidence of 8.7/1000 population (Adak *et al*. 2002), and expressing this as an annual burden of illness in the case study population, results in a DALY score of approximately 0.025 – less than half the score predicted from the flood clean-up process.

Although the estimated number of cases of illness is relatively high, it is likely that many would not be picked up by the routine surveillance system and an elevation in cases of enteric illness due to flooding may not be identified. Under the circumstances it is likely that many people would not visit their GP, or the GP may choose not to take a faecal sample from those reporting, particularly if the worse symptoms have passed. Also, where a sample is taken, the clinical laboratory may not isolate the pathogen causing the reported symptoms. Unless all of these stages are completed the illness will not be entered onto the surveillance system records. Additionally, because of the population dispersion due to flooding, as those affected move out of the area to stay in temporary accommodation or with friends and family, caution is prudent in the interpretation of the partial surveillance data available.

The exploratory QMRA was essentially quite conservative for the type of flooding scenario described, in that no account was taken of people mitigating their exposure (see following section) by wearing gloves, nor was any reduction in pathogen concentration allowed for (i.e. pathogen levels were assumed to stay constant from the point of flooding to the time when the clean-up was conducted). In reality, many pathogen levels decrease relatively rapidly in flood water and so the pathogen concentrations to which people were assumed to be exposed are likely to be a significant overestimate. It does, however, illustrate the potential for a notable burden of gastrointestinal illness associated with flooding.

Mitigating Exposure to Infection

In the UK, the Health Protection Agency (HPA 2008) has issued guidelines on the public health impacts of flooding, including general advice on protecting against infection. Contact with flood waters and sediments may be minimized by the use of protective clothing (waterproof boots and gloves) during the clean-up process. Infection risk may also be reduced by taking general hygiene precautions such as thorough hand washing after contact with either flood waters, sewage or sediments, or items contaminated by these. The risk of wound infection during flooding and remediation can be reduced by keeping open cuts and sores clean and dry where possible, avoiding contact with flood waters, and wearing waterproof plasters.

The risk of exposure of children to flood-associated pathogens (inside and outside the home) can be mitigated by keeping them away from flooded areas and contaminated items, including toys and bedding, until the areas and objects can be thoroughly cleaned, disinfected, dried and returned to their normal condition. Clothes worn during clean-up activities should be washed in hot water, separate from any uncontaminated clothes and linen.

Attention should be paid especially to all areas of the kitchen while cleaning up, to ensure that pathogens are removed from objects and surfaces where food is stored, prepared or served. Food contaminated by flood water should be discarded. Hands should be washed before eating or preparing food.

Conclusions

The estimate derived from the QMRA suggests a significant level of mild gastrointestinal illness (46 cases from a hypothetical population of 1028) as a result of the post-flood clean-up process, especially in relation to the estimated risk of infection from the evacuation process of only two cases of illness (data not shown; see Fewtrell *et al*. 2008a). The results from QMRA are clearly dependent upon the defined flood scenario and some of the assumptions made about flood water contamination and people's behaviour during the clean-up process. It is likely that wearing gloves, for example, during the clean-up process would reduce pathogen exposure and hence the risk of infection. Relatively little, however, has been published on people's behaviour during withdrawal as part of evacuation and the subsequent clean-up process. Research is currently underway to examine this process to allow this to be factored into subsequent QMRAs.

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Part 7
Case Studies

Flood Risk Science and Management
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22 Modelling Concepts and Strategies to Support Integrated Flood Risk Management in Large, Lowland Basins: Río Salado Basin, Argentina

RODO ARADAS, COLIN R. THORNE AND NIGEL WRIGHT

Introduction

Our approach to modelling floods must evolve as our knowledge of the environmental, social and economic dimensions of flooding improves and our appreciation of the need to account for the uncertain impacts of climate and societal changes on future flood risks and their management continues to grow. Integrated Catchment Planning, Integrated Flood Risk Management and the imperative to conserve environmental capital through sustainable use of land and water resources have long been accepted as important concepts by stakeholders involved in the project planning and appraisal cycle (Gardiner 1991; Newson 1997). Indeed, for over a decade, global funding agencies such as the World Bank have explicitly included these concepts in the terms of reference they specify when procuring river basin plans (Serageldin 1994). Increasingly, these concepts are formally reflected in project appraisal mechanisms, so that they guide both the thinking and the practical steps by which stakeholders and scientists make choices between the various options for water resource and flood risk management in large basins. Application of the concepts can be widely observed in flood risk management projects in developing countries where funding agencies ensure that there are clear rules to support where their capital loans are being invested; the Storm Drainage Master Plan for Buenos Aires (Halcrow *et al*. 2001) and the Integrated Master Plan for the Río Salado Catchment (Halcrow Group 1999) are two recent examples of World Bank-funded projects for developing flood management plans for urban and rural areas, respectively, within which the terms of reference express the aspiration for integrated and sustainable approaches. The challenge remains, however, to develop methodologies that are adequate to address the underlying issues within inevitable constraints of cost, time and data availability.

The concept of sustainability in the decision-making process implies the adoption of courses of action that pay due regard to climate change, socioeconomic development and the consequences of decisions made today for future generations, which necessarily implies dealing with considerable and irreducible uncertainty. This is manifest particularly in the case of decision-making with respect to flood risk management strategies, as decisions are necessarily based on the highly complex, interactive and highly uncertain relationships between natural hydrological phenomena, the subtle landscape attributes of the basin and the legacies of past and present human

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interventions in the flooding system. When decisions must be made concerning future management of flood risks, the situation is further complicated by uncertainties concerning the timing and severity of climate change, the existence of multiple alternative scenarios for socioeconomic development, changes in public perceptions and expectations with respect to flooding, and the possibility of technological developments in flood damage mitigation that cannot at present be predicted (Evans *et al*. 2003; Thorne *et al*. 2007). Consequently, integrated flood risk management studies, encompassing the conditional probability of inundation from a variety of flooding mechanisms, risk tolerability [e.g. the ALARP principle promoted by the UK's Health and Safety Executive (HSE 2001)] by individuals and society, and the sustainability of different management strategies, emerge as a vital discipline to support decision-making. For example, increasing awareness of climate change places ever increasing emphasis on dealing with future uncertainties, as demonstrated by the Risk and Decision Making Framework developed by the UK Climate Impacts Programme (UKCIP; Willows and Connell 2003). The iterative and tiered nature of this framework is a reflection of the need for management to be both flexible and adaptable in order that it can deal effectively with unpredictable problems, respond to changing stakeholder wishes, and respond effectively to feedback from the beneficiaries of river basin projects.

In this context, this chapter presents flood risk concepts and modelling techniques that are selected to be particularly suited to applications in large, low-lying basins, and examines these concepts and techniques from the perspective of broad goals for catchment development.

The hydrological attributes inherent to large plains have been extensively treated in a very interesting compilation of studies (UNESCO 1984, 1993) covering conceptual aspects (Fertonani and Prendes 1984; Kovacs 1984; Mull 1984; Paoli and Giacosa 1984) and international case studies, such as the shallow aquifers of East China (Shi and Ke 1993); the Caspian Sea Basin (Velikanov 1993) and the Australian approach to hydrology of lowland plains (Chapman 1984). This compilation also placed special emphasis on the unique features encountered within the plains of Buenos Aires Province, which includes the Río Salado Basin.

The preceding introductory discussion summarizes some key issues related to large lowlands, recognizing that low energy is the main feature responsible for the small horizontal water fluxes and the prevalence of vertical fluxes involving evaporation, infiltration and, in particular, the exchange between the shallow aquifer and the unsaturated zone. This prevalence of vertical over horizontal water movements makes lowlands particularly sensitive to climate change and human interventions in the hydrological system. This sensitivity is clearly demonstrated by the recently increased frequency and duration of flooding in the Northwest area of the Río Salado Basin, which is discussed in detail in later sections.

That introduction (UNESCO 1984) also identifies several fundamental issues that are explored in more detail with a case study of the Río Salado Basin more generally. For example, the multi-disciplinary nature of flooding mechanisms, their modelling and their management in lowland catchments require that groups of specialists work together in developing an integrated, conceptual approach right from the beginning of a study, rather than working in series or in parallel throughout the duration of the assignment, and this is addressed through the Framework for Catchment Modelling Studies (FCMS) proposed in a later section (see 'Modelling framework for flood risk management'). Other issues that are explored include the need to develop broad-scale flood risk maps using remote sensing (UNESCO 1993); the importance of adopting water balance models that clearly identify the dominant, driving variables at a variety of scales; the key role of groundwater–surface water interaction; and, in particular, the role of groundwater storage in acting as the 'memory' of the system through its lagged response to hydroclimatic trends and cycles. The latter is clearly depicted by relatively modern, groundwater-induced flooding in the Northwest area of the Río Salado Basin, which is caused by the increase in mean annual rainfall since the 1970s coupled with a landscape inherited from a former geomorphic regime. The work presented here addresses these issues and illustrates the practical advantages gained by integrating project components concerned with hydrology, geomorphology, remote sensing and modelling to develop the capability to produce broad-scale flood hazard and risk maps, as recommended by UNESCO (1984, 1993).

The need to conserve environmental capital when managing flood risk in lowland catchments is increasingly accepted, and in 2001 the American Society of Civil Engineers (Panigrahi 2001) published a review of groundwater and surface water interaction in lowlands that highlighted the multiple functions of wetlands in attenuating floods, supporting water resources and providing ecological habitats (Carter 1984; Hollis and Acreman 1994; Acreman and Adams 1999). Mortellaro *et al.* (1995) define wetlands as 'those areas that are inundated or saturated by surface and groundwater at a frequency and duration sufficient to support a prevalence of vegetation typically adapted for life in saturated soils.' The Everglades in South Florida, USA, is one of the largest 'regional' wetlands in the world. The area features a flat topography, highly permeable sandy soils, high water tables and a dense network of engineered drainage channels. Due to soil permeability and the shallow aquifer conditions, aquifer levels are highly influenced by rainfall, direct evapotranspiration from the saturated zone and seepage to and from the channels. The Everglades is undergoing a major restoration scheme, designed to promote natural processes and reverse the environmental decline caused by the excessive land drainage works and, as part of this project, significant effort has been devoted to developing coupled models of groundwater and surface water interaction. This chapter acknowledges that effort, but it goes further in exploring which processes need to be coupled and the degree of coupling necessary as functions of the scale and objectives of the investigation, with the FCMS providing a platform to analyse these issues.

Flood risk may be defined as the product of the probability of occurrence of a flood event multiplied by the magnitude of the consequences should it occur. From a modelling perspective, mapping flood risk therefore demands the ability to characterize first the extent of flooding associated with an event of a given return period and, second, the exposure to flood hazard (consequences) of the population, property and infrastructure. The first requirement is usually addressed through the generation of floodwater surface elevations in the fluvial system and their expression as flood maps using a Geographical Information System (GIS). The widely adopted assumption that flood frequency estimates can be an adequate surrogate for the actual frequency of floodplain inundation then allows conversion of these maps of flood extent into maps of flood probability. The accuracy of flood probability mapping has recently been markedly improved through application of two-dimensional (2-D) hydrodynamic models and wider availability of digital elevation models (DEMs) based on LiDAR (light detection and ranging) surveys (Horrit and Pender 2008).

However, mapping the probability of inundation presents new modelling challenges in large, lowland catchments where fluvial processes are less dominant and interaction between groundwater and surface water is in itself an important flooding mechanism. This challenge centres on merging data on both the extent and the probability of flooding associated with flooding mechanisms that drive spatially and temporally distinct flooding systems.

Inundation modelling that can reliably represent both the relevant physical processes and the impacts of proposed engineering interventions is vital to the production of useful flood probability maps. Consequently, this chapter emphasizes the generation of flood probability maps that reliably represent the coincident risks of groundwater and fluvial flooding to support accurate appraisal of alternative strategies for reducing flood risk. However, inundation modelling is particularly challenging when dealing with large basins, complex hydrological processes and a wide array of options for flood risk management. This reinforces the importance of developing a modelling strategy that is able adequately to represent flood processes while also being capable of informing decision-making by being sufficiently fast and flexible to perform the multiple runs necessary to support an integrated appraisal of a range of options for intervention.

The practical utility of the concepts and modelling strategies presented herein is demonstrated using examples and experience gained during the development of a master plan for flood risk management in catchment of the Río Salado, Argentina (Halcrow Group 1999).

The Río Salado Basin

The Río Salado Basin covers an area greater than 170,000 km^2 in Buenos Aires Province, Argentina (Fig. 22.1), and has a population of around 1.3 million. The catchment is one of the most important lowlands in Argentina, with only 130 m of rise in elevation from the sea to the watershed 600 km to the west. The generally low relief is interrupted only by two ranges of Precambrian hills (ascending to about 900 m above sea level) in the far south of the basin. The Río Salado Basin plays a key role in the nation's economy, contributing 25% of grain and 30% of meat production respectively. However, a large proportion of the catchment suffers from frequent and persistent flooding, which severely constrains production and prevents realization of the region's full economic potential. In addition to their impact on the agricultural economy, floods also threaten numerous settlements within the basin, disrupt road and rail communications and put vital infrastructure at risk.

The physical geography of the catchment is the single most important factor affecting the way that floods develop and impact the people, commodities and infrastructure. The landscape of the entire catchment is dominated by ancient aeolian features that have been modified by more recent fluvial processes, reflecting the humid conditions that have developed in what was once an arid region (Fig. 22.2). In essence, relic dune fields dominate the terrain of the Northwest area (Region A in Fig. 22.1), while the landscape of the eastern and central parts of the catchment is characterized by numerous lakes, wetlands and marshes (Region B in Fig. 22.1). The basin's very low slopes, coupled with the relative youth of its surface water drainage system, means that rivers draining the catchment have not yet had time to adjust to the present, more humid climate, and they lack both the drainage density and channel conveyance capacity necessary to convey even

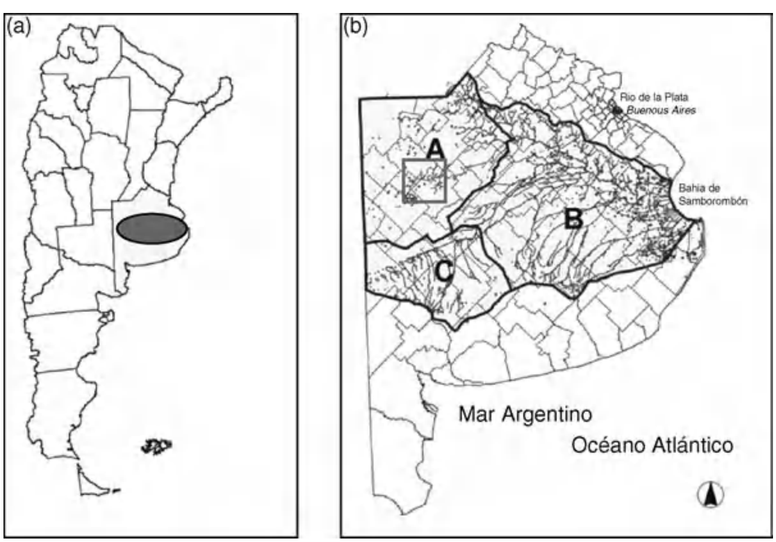


Fig. 22.1 (a) Location of Buenos Aires Province within Argentina. (b) Location of Río Salado Basin within Buenos Aires Province. A: Northwest Region; B: Salado, Vallimanca and Las Flores river valleys; C: South-Western Lake System. The red box shows the location of a dune field area within the Northwest Region (A). (See the colour version of this figure in Colour Plate section.)

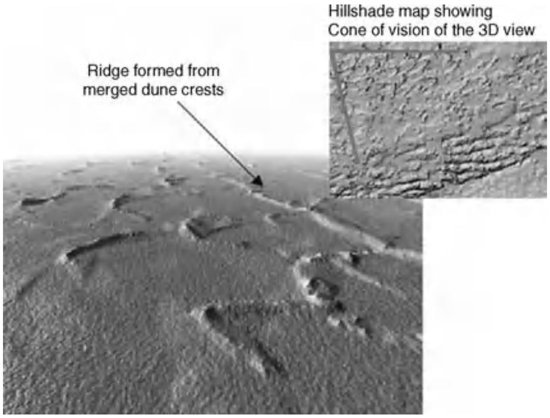


Fig. 22.2 Three-dimensional view of aeolian features in the Northwest of the Buenos Aires Province (red box in Fig. 22.1b). (See the colour version of this figure in Colour Plate section.)

average annual floods within banks. In fact, the basin is so poorly drained by its surface water channel system that the mean flow of the Río Salado close to where it discharges to the ocean is only 73 m^3/s, while its maximum historic flow was measured in 1985 at just 446 m^3/s, both discharges being far below what would be expected for a catchment of this size. The reason these discharges are so small is that excess precipitation is stored through long-term, catchment-wide flooding (Fig. 22.3).

The extent and duration of flooding in the basin has increased during the last four decades, due to increased precipitation that is probably part of a wet cycle associated with natural climatic variability. The impact has been greatest in the Northwest of the basin and, as this subregion has changed from being semi-arid to semi-humid, groundwater levels have risen by up to 7 m and land use has changed from predominantly pasture to arable.

Problems associated with flooding in the basin first began to attract attention at the beginning of the 20th century, leading to several studies (CFI 1980; AEE 1990) of varying scope and geographical extent. It is important to point out that during this period (prior to implementation of artificially engineered drainage works), the Northwest Region (Region A) and the Western lake system (Region C) constituted separate drainage systems that did not connect or interact with the natural fluvial system of the Río Salado (Region B). In 1997, the provincial government commissioned development of an Integrated Master Plan (IMP) to identify possible long-term, basin-wide solutions to flooding and waterlogging problems and, in turn, create more favourable conditions for farming investment and development in the catchment. While the primary aim was to enable farmers to increase their productivity, it was recognized that this plan must be realized within a framework of sustainable management and conservation of environmental capital. The IMP was funded by the World Bank, and the results were presented to the provincial government in December 1999 (Halcrow Group 1999).

Development of the IMP was the first attempt at an integrated approach to managing flooding problems at a regional scale in Argentina, comprising studies across all of the relevant disciplines, from

(a)

(b)

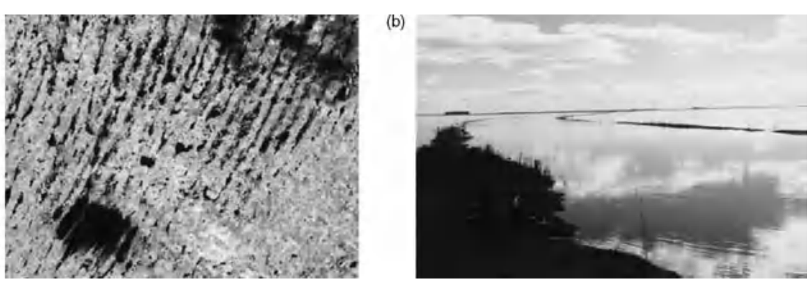


Fig. 22.3 (a) Satellite image showing flooding in the Northwest of Buenos Aires Province (Region A) – see flooding captured behind the dune crests shown in Fig. 22.2. (b) Flooding in the Río Salado (Region B). (See the colour version of this figure in Colour Plate section.)

baseline studies of hydrology, geomorphology and the environment, through complex computer modelling and multi-criteria analyses of alternatives supported by an economic evaluation. Models constituted the key tools employed to gain an understanding of the main flooding and waterlogging mechanisms and, consequently, to assess the potential impacts of a range of proposed strategies for flood risk reduction.

Flooding Mechanisms and Risks in Large Lowland Basins

General concepts

The first step in any flood risk analysis is to characterize the flood probability by understanding the mechanisms responsible for flooding. Some of the prime considerations in large lowland basins include:

1 Scale: The physical size and complexity of large basins presents challenges to modelling and management strategies in terms of data needs and availability, the resolution required and the appropriateness of different analytical tools (bearing in mind modelling costs and run time constraints).

2 Catchment and study area boundaries: When dealing with large national or trans-national catchments more than one administrative body will usually be responsible for catchment and flood risk management. This poses challenges in terms of procuring information and dealing with multiple institutional arrangements and, hence, the possibility of multiple preferred policies and options for managing flood risk.

3 Geomorphology: In large, low-lying catchments the importance of geomorphic processes and features is enhanced as complex interactions between the hydrology and basin geology, soils, vegetation and landscape control not only the generation of runoff but also how, when and where waterlogging and groundwater flooding occur.

4 Enhanced role of interaction between surface water and shallow groundwater processes: In lowland catchments vertical fluxes and shallow processes predominate due to the lack of relief. Particularly important processes include: extensive evapotranspiration from the subsurface saturated zone; widespread soil saturation with exfiltration and large-scale ponding of water on the surface; highly variable contributing areas with respect to saturation overland flow and surface runoff; slow movement of underground water leading to saline and brackish conditions; and very low base flows in drainage channels. As will be demonstrated throughout this chapter, groundwater and surface water interaction is a key process responsible for explaining flooding mechanisms in the Río Salado, as well as many other large, lowlying basins.

5 Thresholds: In relation to points 3 and 4 above, geomorphic features may generate threshold behaviour in surface water–shallow groundwater interactions, triggering significant step changes in the depth and extent of surface flooding. See Figure 22.7 for an example of this phenomenon.

6 Engineering interventions: Finally, engineering interventions in large, low-lying basins require particularly careful and thoughtful planning supported by broad-scale scientific studies. This is the case because the flat relief and slow lateral movement of water in large, lowland areas means that the terrestrial hydrological system responds slowly to changes in precipitation and the operation of structural flood management structures, making it necessary to anticipate the natural, autonomous evolution of a flood event in order to intervene and manage it effectively. For example, in the Salado basin, floods develop over months rather than days or weeks, and forward planning is essential to operate sustainable land drainage infrastructure (regulated drainage canals) to mitigate groundwater-induced surface flooding while simultaneously avoiding unacceptable damage to the environment.

Given the complexities inherent to the considerations outlined above, risk management provides the logical framework within which to assess current and possible future flooding scenarios that account for various options for water resource management and different trajectories of socioeconomic evolution.

Flood risk depends on the probability of flooding and the consequences should a flood actually occur. Hence, when assessing flood risk it should be noted that the consequences of a flood do not share the boundaries defined by the duration and extent of the inundation. For example, with respect to agricultural and livestock activities, adverse effects extend beyond the inundated area because disruption of the transportation network may prevent the movement of animals or their feedstuffs, while the duration of adverse impacts usually lasts much longer than the time necessary for soil moisture to return to normal levels. Under-representing the extent or duration of the adverse consequences of flooding leads to under-estimation of flood losses, which may result in faulty decisions on flood risk management and, hence, poor use of resources.

To understand how flood probabilities, hazards and consequences are linked, and so assess the potential for different mitigation measures to reduce flood risk, conceptual models of risk such as 'Source-Pathway-Receptor' (SPR) have proven useful (Fleming 2002). Their application provides the basis for representations of flood probabilities, hazards, exposures and, ultimately, risks that are accurate and consistent across regions that are both large and complex. However, to produce reliable estimates of current and future flood risks, models such as SPR require a thorough understanding of the physical sources of floodwater, the pathways by which floodwaters move through the basin, and the socioeconomic factors that influence the sensitivity or resilience of a flood-prone area to suffering losses. Once developed and validated, conceptual models of flood risk such as SPR can also be useful to inform decision-making on when and where to deploy flood risk management measures. While little can usually be done to manage the sources of floodwaters, flood defences may be positioned to block key flooding pathways, while non-structural measures such as floodplain zoning and land use planning to reduce exposure are powerful responses aimed at reducing the adverse consequences of flooding.

Taking into consideration all of the issues highlighted above, three types of hazardous flood events can be conceptualized in large, lowland basins:

1 **Groundwater-induced surface flooding** which occurs when the groundwater level (phreatic surface) rises to an elevation above the ground surface, causing water ponding and inundation, or **groundwater-induced waterlogging** which occurs when the phreatic surface is located within 50–70 cm of the ground surface, but does not rise above it.
2 **Ponding of rainwater** which occurs when water accumulates on the surface due to low infiltration capacity and/or flat terrain.
3 **Fluvial flooding** which occurs when the discharge of water from upstream exceeds the conveyance capacity of the natural or engineered channel and, consequently, a significant area of the surrounding floodplain is inundated.

Box 22.1 describes how a conceptual SPR model can be applied to characterize the resulting risk in a basin dominated by groundwater-induced flooding mechanisms.

It is necessary to understand flood risk conceptually, and define adequate risk models, prior to the delineation of the strategy for model development. This is the case because it enables identification of the analytical components that must be included in the modelling effort, together with the degree of coupling required between models to adequately capture interactions between the various flooding sources and pathways that together control the conditional probability that a given area may be flooded. In this respect, the interactions likely to occur in large, lowland catchments operate at a wide range of scales and this must be reflected in their analysis. Figure 22.4 illustrates possible interrelationships at local and regional scales. For instance, groundwater-induced flooding can interact with rainfall to cause a further increase of water ponding on the surface and, hence, an increase of overland flow that promotes fluvial flooding.

To flesh out the modelling strategy presented in Figure 22.4, it is next necessary to conceptualize how the three flooding mechanisms outlined above act and interact physically. Figure 22.5 presents a conceptual representation of the response of the basin to each flooding mechanism, illustrating the importance of thresholds in generating non-linear behaviour in the flooding system.

Box 22.1 Application of Source-Pathway-Receptor model to groundwater-induced flooding in the Río Salado Basin

Groundwater-induced waterlogging

Source
The primary source of this type of flood risk is groundwater in areas where the phreatic surface is within 50–70 cm of the land surface.

Due to the relatively high infiltration capacity of the sandy soils that are prevalent in Region A (Fig. 22.1b and Fig. 22.2) and the weak horizontal dynamics of the groundwater system, it may be argued that the propensity for rainwater to infiltrate vertically and accumulate in the soils and so increase the groundwater level constitutes a secondary source of this source of flood risk.

Pathway
This risk exists predominantly throughout the Northwest area of the basin (Region A), where a delicate balance between rainfall and evapotranspiration – coupled with the geomorphological characteristics of the area – maintains groundwater heads very close to the land surface.

Receptor
The main receptor of this flood hazard is agricultural activity, which is adversely affected through:

- salinization of soils due to the rise of groundwater with a high salt content;
- physical limitations on the growth of crops during the initial stage after sowing;
- difficulty of accessing the land for harvesting.

Groundwater-induced surface flooding

Source
The primary source of this type of flood risk is water that exfiltrates in an area where the groundwater table is at or above the land surface. This type of flooding also leads to ponding of rainwater during precipitation events.

Pathway
This risk also exists throughout the Northwest area of the basin (Region A), where no surface water drainage network has yet developed. Groundwater-induced surface flooding is common along interdunal depressions. It appears first on the upstream side of parabolic dunes within the interdunal depressions. Initially, the flooding so caused is not harmful. However, when the flood depth increases sufficiently that water overtops the crests of the parabolic dunes, surface water cascades downstream along the interdunal depressions, creating substantial volumes of surface runoff and forming a significant pathway for this flooding mechanism.

Receptor
The receptors of this process are also the agricultural activities in the Northwest region. Losses occur through disruption of livestock activities (unlike waterlogging) and farm infrastructure (such as fences).

The occurrence of surface water flooding also implies crop damage. However, this is a relatively small loss as cropping is not a major activity in these low-lying areas within the region.

Flooding mechanisms in the Río Salado basin

Fluvial flooding occurs frequently in the Río Salado Basin because the drainage system has not yet adjusted to the current runoff regime. In fact, the drainage system is formed by under-fit channels superimposed on relief formed by aeolian (wind-blown) landforms that constrain the

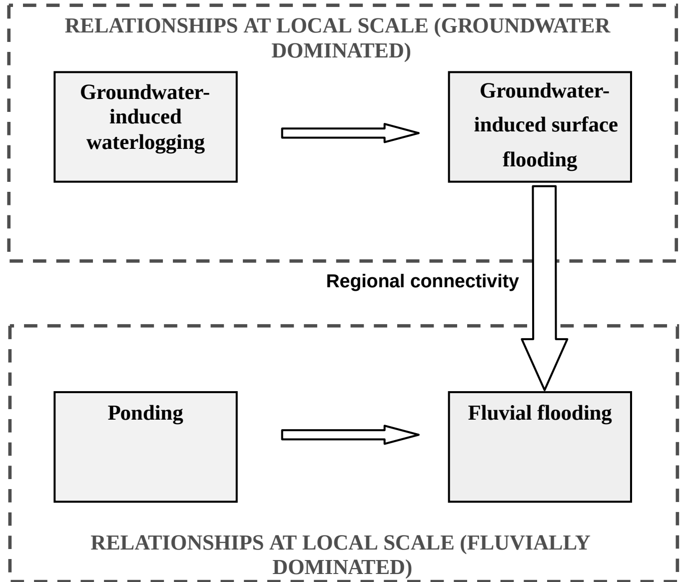


Fig. 22.4 Interrelationship between flood mechanisms at local and regional scales.

conveyance of both surface water (groundwater-induced) and fluvial (out-of-bank) floodwaters. In the lower Río Salado (Region B), topographic constrictions on the channel formed by relic aeolian ridges separating deflation hollows are particularly strong and come into operation earlier in the flooding process, limiting the egress of water from the channel, causing important backwater effects and delaying the downstream conveyance of floodwaters. Figure 22.6 shows a satellite image of the lower reach of the Río Salado and a typical cross-section featuring a topographic constriction.

Groundwater-induced surface flooding is a direct result of interaction between the regional hydrogeological system and microforms in the surface relief. For regional hydrogeological purposes, the system can be considered as unconfined, with its base defined at the top of the Parana Formation. Apart from the fringing mountain and hill areas at the boundaries of the basin, groundwater gradients are very gentle, with a regional flow from west to east. The ubiquitous presence of wetlands and lakes (progressively increasing in frequency from west to east), is dictated by the very gentle topographical gradient and very shallow unsaturated zone, and indicates that the groundwater regime is closely coupled with, and constrained by, the surface water regime. Geomorphologically, a feature of the basin is the lack of natural channels acting as tributaries to the principal rivers and arroyos. This shows that a conventional surface water drainage system with significant direct runoff from rainfall is not dominant in the Northwest (Region A).

Because of the shallow depth of the unsaturated zone, the water table is directly subject to evapotranspiration. When major recharge events occur, normal evapotranspirational losses are overtaken, heads rise rapidly and exfiltration takes place, giving rise to flooding and the subsequent creation of non-perennial surface water bodies and areal extension of perennial lakes. Depending upon their connectivity (controlled by topographic thresholds such as dune crests) and the amount of exfiltration, these lakes can join to form cascades and major groundwater-induced flooding can ensue. Consequently, surface flooding induced by seasonal and event-driven accumulation of groundwater is a transient phenomenon. During dry periods the groundwater system is localized, with recharge being largely balanced by evapotranspiration, resulting in limited lateral groundwater contributions to regional runoff. The result is that regional runoff is relatively small,

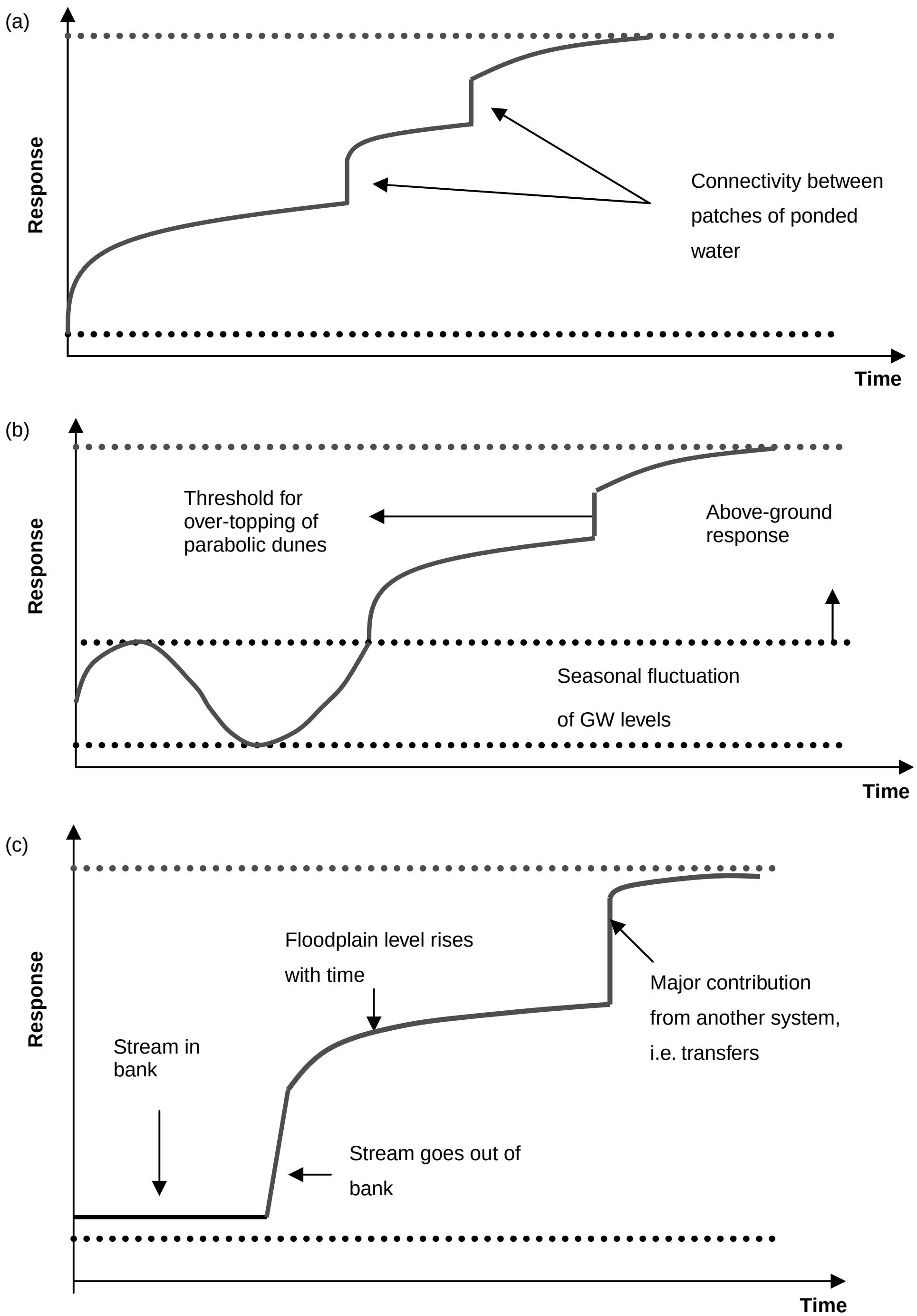


Fig. 22.5 Conceptual diagrams of the response of a basin to flooding. (a) Ponding of water; (b) groundwater-induced flooding and (c) fluvial flooding (dotted red and blue lines: maximum and minimum responses, respectively). GW, groundwater.

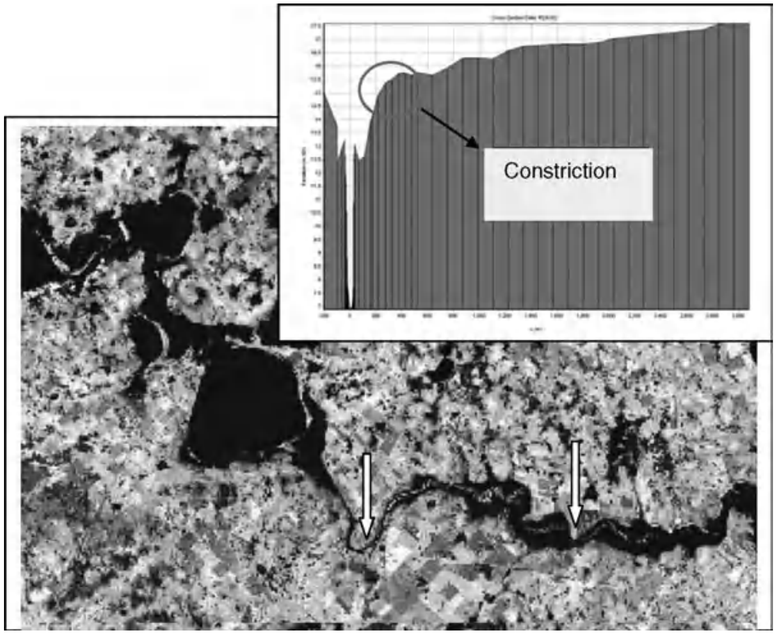


Fig. 22.6 Interaction between relict aeolian features and fluvial flooding in the Lower Salado. (See the colour version of this figure in Colour Plate section.)

which is consistent with the relatively low hydraulic conductivity, the very gentle regional gradients and the relative high groundwater salinity. Figure 22.7 shows a conceptual representation of the groundwater-induced surface flooding process.

Based upon this conceptual understanding of the surface water–groundwater system, the flow balance for the basin can be formulated as follows:

Precipitation = runoff + infiltration + interception storage + evaporation from open surfaces with the intercepted water

Infiltration = actual evapotranspiration from the unsaturated zone + lateral subsurface flow (interflow) + recharge

Change in groundwater storage = recharge ± groundwater flow − actual evapotranspiration from the saturated zone ± groundwater discharge to rivers, wetlands and lakes

Approaches and Techniques: The Role of Mathematical Modelling

Approaches and techniques

Having conceptualized a flood risk model that encompasses the various flooding mechanisms operating in a large, lowland catchment, the next step is to develop a modelling approach that accurately transfers the embedded concepts into a practical assessment of current and future flood probabilities and risks. A generic approach (successfully applied to the Río Salado Basin) is proposed in Figure 22.8. In this figure, parallels are set between the conceptual understanding of the processes that lead to flooding, the tools required to simulate these processes and the components of the flood risk model.

A key output of the modelling approach is the Flood Probability Map (FPM). The FPM is a spatial representation of the probability of flooding associated with events of given magnitudes. This is often expressed in terms of the return period of an event, i.e. the inverse of the annual frequency of exceedence of that event. By definition, an FPM

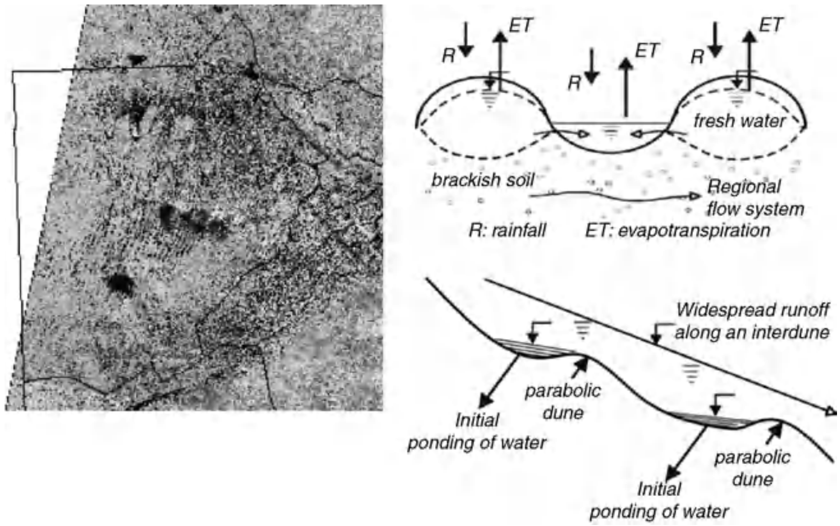


Fig. 22.7 Conceptual representation of groundwater-induced surface flooding in the relict dune field of the Northwest part of the Río Salado Basin (Region A). (See the colour version of this figure in Colour Plate section.)

Processes	Climatic stresses and hydrological processes	Geomorphic setting and hydraulic infrastructure	Human activities and social, environmental and physical infrastructure	Impact on social, environmental and physical capitals

(damage curves)

Tools	Mathematical models	Digital Elevation Model	Social and agro-economic databases
	Flood Probability Map (FPM)		Flood Risk Map (FRM)

Risk Elements	Source	Pathway	Receptor

Fig. 22.8 Generic approach to estimating flood risk.

encompasses representation of the following elements:

1 The spatial distribution of hydrometeorological stresses (rainfall, evapotranspiration).

2 The response of the physical environment (soil layer, aquifer, fluvial corridor) to the hydrometeorological stresses.

3 The spatial representation of hazard; e.g. flooding/no flooding and waterlogging/no waterlogging situations in the basin.

4 The calculation of the probability of occurrence of a particular event (groundwater-induced flooding, waterlogging, fluvial flooding).

Mathematical models provide the basis for generation of FPMs and, in this context, it is important to note that while the first element (point 1 above) is a direct function of the availability of temporal and spatial records of precipitation and evapotranspiration, the other three elements are functions of the modelling strategy adopted for the studies.

The FPM yields, for every discrete element (raster element) of an area, the number of times during a specified period that the water level is expected to rise to a given elevation above ground level (for groundwater-induced surface and fluvial flooding) or just below it (for groundwater-induced waterlogging). The general approach to calculating the flood frequency is set out in Box 22.2.

Modelling framework for flood risk management

The importance of carrying out integrated studies and developing flood risk management plans at the basin scale when seeking sustainable solutions to flooding problems is now widely recognized. However, the requirement to generate and incorporate multiple inputs from a range of cognate disciplines leads inevitably to the development of ever more complex mathematical models that seek to simulate as closely as possible the physical processes that occur in nature. This makes it increasingly important that a sound methodological platform is used to underpin selection of the appropriate modelling approaches, tools and linkages. This is necessary in order that the methodological platform:

1 supplies the deliverables that are actually required to support integrated catchment studies to guide the development process;

2 ensures that the model outputs are compatible with the requirements of decision-makers not only at the end but also at each intermediate stage of the study;

3 avoids expending monetary and human resources unnecessarily.

Current flood risk management policies demand an integrated modelling platform or framework, but despite this there remains an absence of a formalized modelling framework. Aradas (2001) proposed a Framework for Catchment Modelling Studies (FCMS) as 'a staged and systematic approach to be used as a template for the development of modelling exercises to suit the physical characteristics of a basin and the level of detail required at each stage of a project, trying to strike a balance between project needs, cost and human resources.'

The resulting FCMS recognizes three distinct blocks of activities, coinciding with the three stages proposed by Mitchel (1989), to guarantee a timely and cost-effective project planning process. The blocks are:

1 Normative level: identifying and examining general issues regarding the hydrological, geomorphological, land-use and flooding systems in the basin.

2 Strategic level: comprehensive analysis of land and water interactions at a variety of scales throughout the basin.

3 Operational level: integrated modelling of selected, key flooding mechanisms in the basin.

The proposed FCMS (Fig. 22.9) parallels existing approaches to geomorphic studies, environmental assessments and engineering project management (Thorne 1998). Indeed, one of the lessons learned from research performed to develop the FCMS was the need to better account for geomorphic processes and landforms in the mathematical modelling. For example, in the Río Salado Basin it emerged that the proper representation of the hydrological impacts of the different types of relict dune field present in the Northwest (Region A) is an essential

Box 22.2 Approaches for dealing with the calculation of flood frequency

The general statistical expression for the calculation of the frequency of occurrence of an event is:

$$Frequency_i = \frac{NoEvents}{(n+1)}$$

where:
i = point in space for which the probability is calculated;
$NoEvents$ = total number of events in the period of analysis (n).

In terms of modelling, two approaches to calculating the probability of occurrence of an event can be identified, as follows:

Approach I

$$F_{event} = \Phi[Frequency(Input)]$$

Approach II

$$F_{event} = Frequency[\Phi(Input)]$$

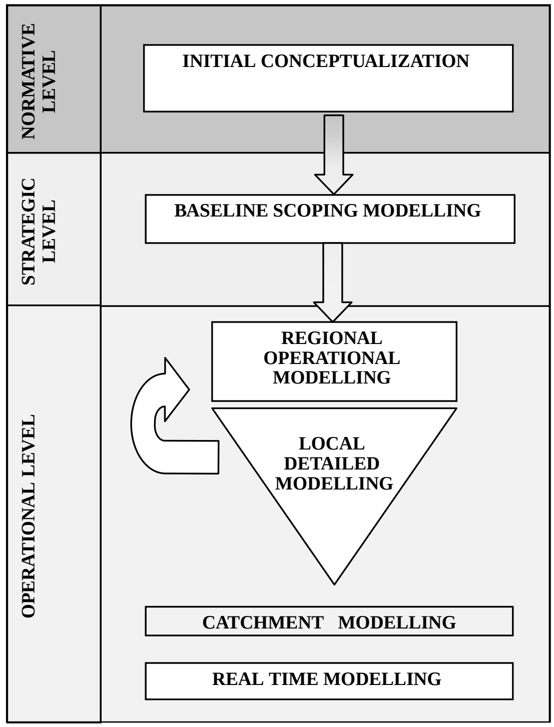
where:
F_{event} = frequency of an event, e.g. the occurrence of groundwater-induced surface flooding, expressed as the number of times over a period that water is above the ground surface;
Input represents the driving hydrometeorological phenomenon, e.g. a flow hydrograph or a time series of rainfall, and
Φ represents the response function of the physical system to the hydrometeorological driver.

The first approach is usually emloyed in the representation of fluvial flooding, where the above expression translates into the following steps:

1 Statistical analysis of annual series of flow maxima to derive a time series (flow-time; rainfall-time) with different frequency of occurrence [***frequency(****input)*].
2 Execution of a mathematical model (i.e. a 1-D river flow model) using the above inputs to estimate the response of the system in the form of predicted water levels along a river and its floodplain [Φ*(frequency(input))*].
3 Calculation of the flood extent associated with the water levels for each return period (yielding the **FPM**).

The second approach is most suitable for the representation of the occurrence of distributed events and is appropriate for groundwater-induced flooding processes. In this case, the approach normally consists of the following steps:

1 Execution of a mathematical model (i.e. a hydrological rainfall-runoff model coupled to a 3-D groundwater flow model) using time series of rainfall and evapotranspiration (*input*) to estimate the groundwater level for every simulated time step over the period of calculation (*n*). This results in the calculation of [Φ*(input)*]:
2 Calculation of the number of occurrences of a flood event, performing a calculation of a flood/no flood situation for every time step by comparing the groundwater level from the model with the level of the ground surface.
3 Calculation of the flooding frequency as [*frequency(*Φ*(input))*].



FRAMEWORK FOR GEOMORPHIC STUDIES	FRAMEWORK FOR CATCHMENT MODELLING STUDIES
PERFORMANCE CRITERIA	INITIAL CONCEPTUALIZATION
CATCHMENT BASELINE GEOMORPHOLOGY	BASELINE SCOPING MODELLING
FLUVIAL AUDIT	REGIONAL OPERATIONAL MODELLING
FLUVIAL DYNAMICS ASSESSMENT AND SUSTAINABLE DESIGNS	LOCAL MODELLING
GEOMORPHOLOGICAL INPUT TO WATER RESOURCES MANAGEMENT	CATCHMENT MODELLING
POST PROJECT APPRAISAL	REAL TIME MODELLING

Fig. 22.9 Proposed Framework for Catchment Modelling Studies (FCMS) and its parallel with geomorphic studies.

prerequisite to understanding and accurately simulating the groundwater–surface water interactions that drive the flooding system in the modelled test area.

Modelling Groundwater and Surface Water Interaction

This section focuses specifically on the flooding mechanism that is dominant in lowland basins: groundwater–surface water (GW–SW) interaction (Mull 1984; Paoli and Giacosa 1984; Sacks *et al*. 1991; Aradas 2001), as described in Box 22.3. The importance of this mechanism goes beyond flood risk management as it also has implications for water resources management and the conservation of key environmental assets such as wetlands and marshes. Modelling GW–SW interaction poses very distinct challenges in relation to issues of temporal and spatial scale. This is particularly the case when the model outputs are required not only to simulate hydrological processes per se but also to provide the information needed to produce key decision-making tools such as flood probability maps.

In the Río Salado studies, two distinct modelling scales were adopted to support development of flood probability maps. Both included appropriate representation of GW–SW interaction:

- A regional model and a regional flood probability map to support the broad-scale representation and validation of flooding mechanisms and to identify priority areas for flood risk management.
- A local model to test the specific (small-scale) impacts of alternative flood risk management measures and support engineering design of those measures selected for feasibility study; for example, the construction of drainage canals to reduce flood probabilities and hence improve agricultural productivity.

It was recognized from the outset that these models would be used in tandem with, for example, some of the results generated by the local model (such as predicted discharges in proposed drainage canals) being fed back into the regional model to test the effectiveness of the local flood risk reduction measures at the regional scale.

Regional modelling approach

Modelling at the regional scale required the use of process-based mathematical models selected to represent the relevant flooding mechanisms, with a simplified degree of coupling. The models selected were:

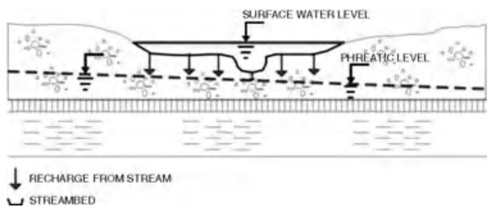
1 HYSIM (Manley 1993: a physically based, lumped, rainfall-runoff model. HYSIM is a deterministic, continuous simulation model that performs a water balance on a lumped basis, taking into account the most relevant hydrological processes including interception storage, evapotranspiration, surface and subsurface runoff, soil moisture content, recharge to groundwater and discharges to watercourses.:

2 MODFLOW: a computer code that resolves the equations for three-dimensional (3-D) flow in a saturated porous medium, using a finite difference scheme (McDonald and Harbaugh 1988). The program deals with recharge and evapotranspiration as external fluxes, in the same way that it deals with external leakage to and from surface water bodies.:

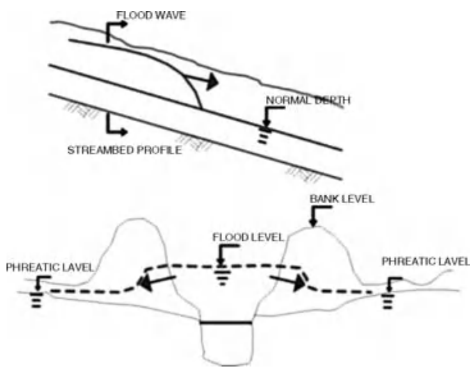
3 iSIS Flow: a one-dimensional hydrodynamic model that solves the St-Venant equations for open channel flow through a mixed system of channels and hydraulic structures (Halcrow Group 2009).

The previous section described the conceptual basis for the flood probability maps (FPMs). At a broad scale, the regional groundwater model was used to compute groundwater heads (from a time series between 1963 and 1995) that, once combined with the Digital Terrain Model, permitted identification of areas of groundwater-induced flooding and waterlogging. ISIS Flow was then used in an uncoupled manner, to generate flood extents for various return periods along the river corridors. Fluvial and groundwater-based FPMs were merged by choosing the lowest frequency of occurrence for those pixels where the predicted flood extents overlapped. A sample FPM, produced using the regional model, is shown in Figure 22.10.

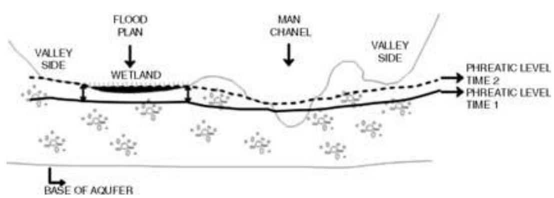
Box 22.3 Typical examples of groundwater–surface water (GW–SW) interaction in lowland basins



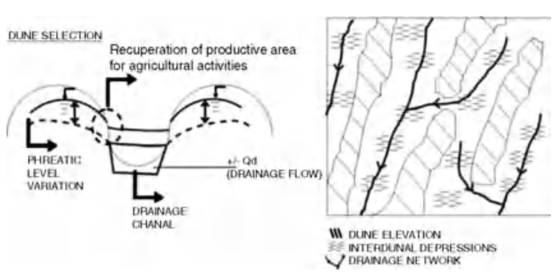
Recharge from leaky streams to an aquifer. Note variation of streambed thickness between the main channel and the floodplain.



Bank storage attenuation effects when a flood wave passes along a stream.



Seasonal water level fluctuations beneath the floodplain of a river, driving changes in wetland hydrology and diversification of habitats, especially at water margins.



Coupling between surface water, groundwater processes and artificial drainage channels is a key factor that determines flow to drains and efficiency of the drains in terms of water table drawdown. This must be understood and accounted for in the engineering design and operational rules for artificial drainage systems.

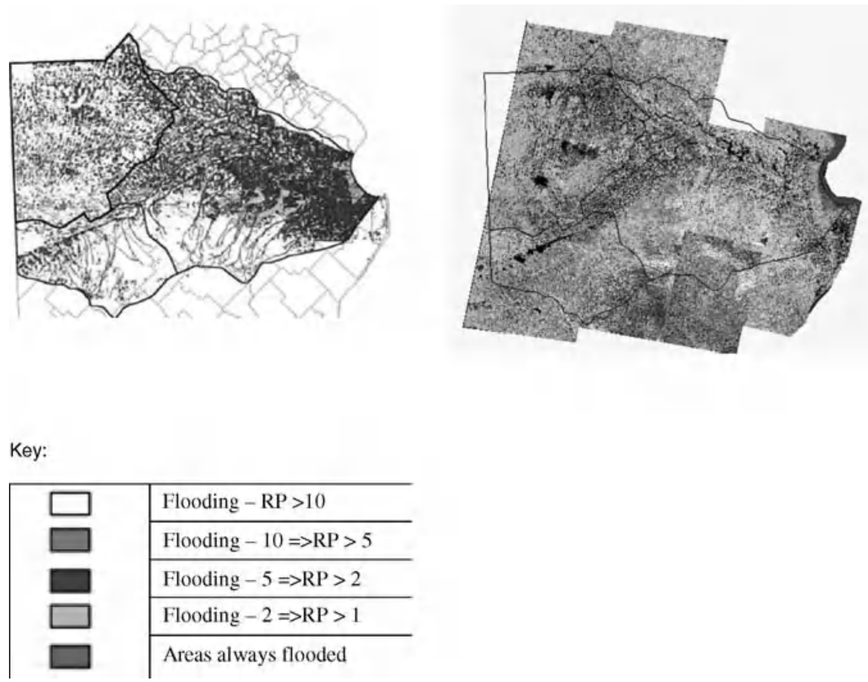


Fig. 22.10 Sample Flood Probability Map (FPM) produced using the regional flood model. RP, return period. (See the colour version of this figure in Colour Plate section.)

Using MODFLOW alone to generate FPMs would not, however, be a robust approach as it can only identify the extent of surface flooding approximately. This is the case because it fails correctly to estimate the depths of water ponded and flowing on the surface. Also, its utility for mapping the probability of surface flooding is tightly linked to a number of factors concerning the schematization of the model. Forcing the resolution of the groundwater model to match the resolution of the micro-relief features helps to identify more precisely the location of flooded areas. However, the fact that model outputs vary with the temporal resolution of the hydrological stresses input to the model (monthly or daily), represents a significant weakness in this approach.

The groundwater heads predicted by MODFLOW are strongly influenced by the vertical components of the water balance, evapotranspiration from the saturated zone and, to a lesser degree, the elevation of the ground surface. This is the case because the fact that horizontal flow is negligible makes the modelled groundwater profile more sensitive to changes in the input stresses and less sensitive to trends in the elevation of the ground surface.

It must be concluded that the utility of this approach as a means of estimating which areas would benefit from construction of drainage channels is limited for the reasons explained above. Further, the only viable method of simulating the effect of proposed drainage channels would be to modify the time series of infiltration as it would be incorrect to include the canals explicitly in the model, because MODFLOW is unable to calculate the correct head of water at the surface to drive water into the drains.

Local modelling approach

Local modelling was also undertaken for a test area in the Northwest part of the Salado Basin (Region A). For this application a coupled model was developed by linking an iSIS model for surface flows with a MODFLOW groundwater model (iSISMOD – see Box 22.4). The aim was to assess the advantages and disadvantages of using a coupled model to generate FPMs, in comparison with the uncoupled modelling approach used in the regional (broad-scale) approach. Predicted surface water levels along longitudinal and cross-sectional profiles within the relict dune fields were examined to compare the model outputs with the actual flooding patterns observed from satellite imagery.

Schematization of the local, iSISMOD model, consisted of two layers: a groundwater component (layer 2) and a surface water component (layer 1). Layer 2 was discretized in the horizontal plane using cells of 500×500 m, in place of the coarser 5000-m grid used in the regional model. Layer 1 of the model was discretized using the units available in iSIS Flow in order to simulate as closely as possible the flooding mechanisms that operate at the surface, including those related directly to the occurrence of groundwater-induced flooding and waterlogging. These mechanisms are:

1 **Exfiltration** of water at the surface, due to groundwater phreatic levels that exceed the land surface elevation.

2 **Ponding** of exfiltrated water behind parabolic dunes and/or spread of water over low areas in the inter-dunal troughs between large, longitudinal dunes.

3 **Transfer** of water from one low area to the next down slope, once the topographic threshold created by the crest of the intervening parabolic dune is exceeded, creating a cascade effect and, occasionally, a major stream of floodwater.

4 **Transfer** of water from low areas to existing drainage infrastructure (canals) and flooding due to insufficient **conveyance capacity** in the canal system.

5 **Saturation overland flow** as rainfall falls onto a variable area of saturated ground.

6 **Seasonal expansion/contraction** of open water bodies, whose areas vary dynamically as a function of rainfall and evaporation.

Points 1, 5 and 6 above illustrate the concept of variable contributing areas for surface runoff, which stems directly from GW–SW interaction and is extremely important to the distribution and severity of flooding, not only in the Northwest region of this basin but also in all other lowland basins.

It was identified in the regional modelling study that the key shortcoming of the uncoupled approach was its inability to reproduce the dynamics of surface water storage and conveyance, once groundwater had exfiltrated. This indicated the need to include within iSISMOD some representation of direct rainfall and evaporation from saturated areas of ground. Figure 22.11 shows a conceptual schematization of the functioning of this feature of the model:

1 iSISMOD detects whether there is a situation of *Flooding* or *No Flooding*, depending on whether the water level in layer 1 is above or below the ground surface. This check is performed for every cell of the model, at every time step in the surface water simulation.

2 For those cells where *Flooding* is detected, the MODFLOW sub-model within iSISMOD does not include infiltration in the continuity equation for layer 2, but only the leakage originating from layer 1. At the same time, the continuity equation for layer 1 will include terms for both rainfall and evaporation.

3 For those cells where *No Flooding* is detected, MODFLOW takes the infiltration or recharge term from a hydrological model (i.e. HYSIM in the case of the Río Salado study) and uses it directly as an input stress to the aquifer.

The iSISMOD model was validated using runs performed for a 33-year period of record and the validated model was then used to generate FPMs for the test area based on rainfall inputs for the wettest period experienced in the test area region (1985–1990). Runs were performed first for the baseline condition and then with proposed flood risk reduction measures in place.

The key question is whether the FPMs generated using the fully coupled GW–SW model

Box 22.4 ISISMOD – A coupled groundwater–surface water (GW–SW) model

ISISMOD is a fully coupled GW and SW model based on combining existing iSIS Flow and MODFLOW models. Coupling between the programs is based on the following concept.

MODFLOW solves the groundwater flow equation and provides the groundwater heads that iSIS uses to calculate the corresponding leakage, based on the water level of the surface water body. The leakage is exported back to MODFLOW and the groundwater heads are recomputed. Iteration between the programs ceases when the difference between two successive leakage volume calculations is less than a specified tolerance. The issue of the different time steps, normally required for the simulation of both systems independently, was addressed by calculating (in iSIS) an average leakage term during a MODFLOW time step, using groundwater heads linearly interpolated every iSIS time step.

The governing equations for aquifer and surface water interaction are described below. The general expression for the leakage between a surface water body and an aquifer can be expressed as follows:

$$Ql = \frac{ks}{t} \times L_{area} \times (hs - HGW)$$

where:
Ql = leakage flow that passes between a water body and an aquifer (L^3T^{-1});
k_s= vertical hydraulic conductivity of the streambed of the water body (LT^{-1}).
t= thickness of the streambed (L);
L_{area}= cross-sectional area normal to the leakage flow direction (L^2);
h_s= water elevation of the surface water body (L);
HGW= groundwater head on the aquifer in correspondence with the surface water body (L).

The resulting continuity equation for an open channel, including a leakage contribution from an aquifer, can be expressed as:

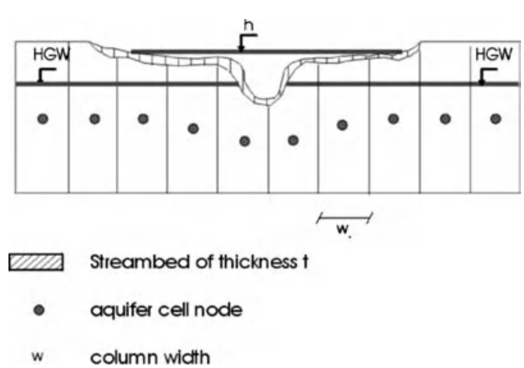
$$\frac{\delta Q}{\delta x} + \frac{\delta h}{\delta t} + ql = 0$$

where:

$$ql = \frac{kv \times L_{area} \times (hs - HGW)}{dx}$$

ql = leakage per unit length of channel ($L^3T^{-1}L^{-1}$).

iSISMOD was developed specifically to allow a wide range of interaction cases, including a large floodplain area interacting with multiple flow cells, as depicted below:



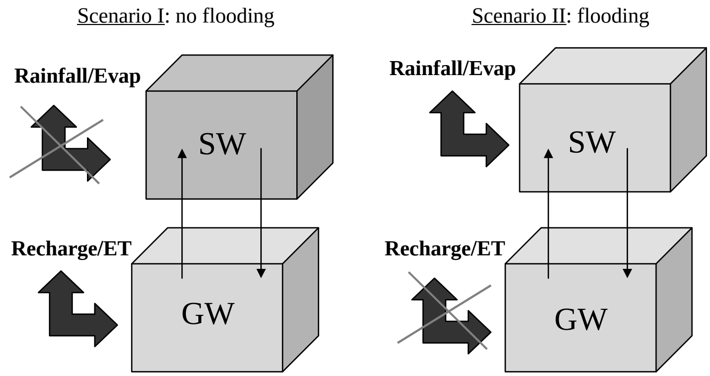


Fig. 22.11 Representation of the simulation of saturated overland flow in iSISMOD. Evap., evaporation; ET, evapotranspiration; GW, groundwater; SW, surface water.

(iSISMOD) in the local approach are better than those generated using the uncoupled models in regional approach. Figure 22.12 allows direct comparison of the FPMs obtained using the local and regional approaches. Comparison reveals that application of MODFLOW at a regional scale, with a grid size of 5000 m (Fig. 22.12a), produces a poor representation of flood probability. Specifically, it under-predicts flooding in both extent and frequency, which is to be expected given the coarse representation of geomorphic landform features provided by the 5000-m grid and the inability of the model to reproduce the dynamics of water at the surface. The finer resolution of the 500-m grid (Fig. 22.12b) improves the identification of areas subject to groundwater-induced flooding markedly.

However, comparison with the FPM generated using the fully coupled GW–SW model (Fig. 22.12c) shows that this is also superior in several other regards:

1 The predicted flooding has increased throughout the test area in extent and frequency and better matches observed patterns. The improvement is particularly strong for the extent of areas flooded during a 2-year return period event.

2 The coupled model accurately detects the change in flood probability in low-lying, downstream areas, which is the logical consequence of these areas receiving surface floodwaters from upstream areas.

3 The appearance of flooding in low-lying, downstream areas (which was not depicted in the regional modelling) for events of low probability of occurrence is also an improvement emerging from the fact that iSISMOD correctly simulates what happens when surface water elevations exceed the threshold set by the crest elevations of relict dune features in the micro-relief along the troughs between the major dunes.

The improved FPMs produced through application of iSISMOD suggest that coupled GW–SW modelling provides the necessary basis for the generation of realistic, physically based predictions of flood probability, although limitations inherent to FPM methodology mask some of the advantages over using MODFLOW alone. Specifically, the fact that the maps are based on calculation of the annual probability of the occurrence of flooding means that the FPMs are rather insensitive to the modelling approach adopted. In fact, this was one of their strengths in terms of the studies used to support the IMP for the Río Salado basin, as it made the results extremely robust.

However, the annual series of flood probabilities is not the best vehicle for demonstrating the benefits of more physically based, coupled models of flooding mechanisms in the generation of flood probability maps. The benefits of improved modelling would be better expressed for flood mapping and management purposes if, in the flood probability analysis, the annual series were replaced by a peak-over-threshold record, based on observations of each flood occurrence.

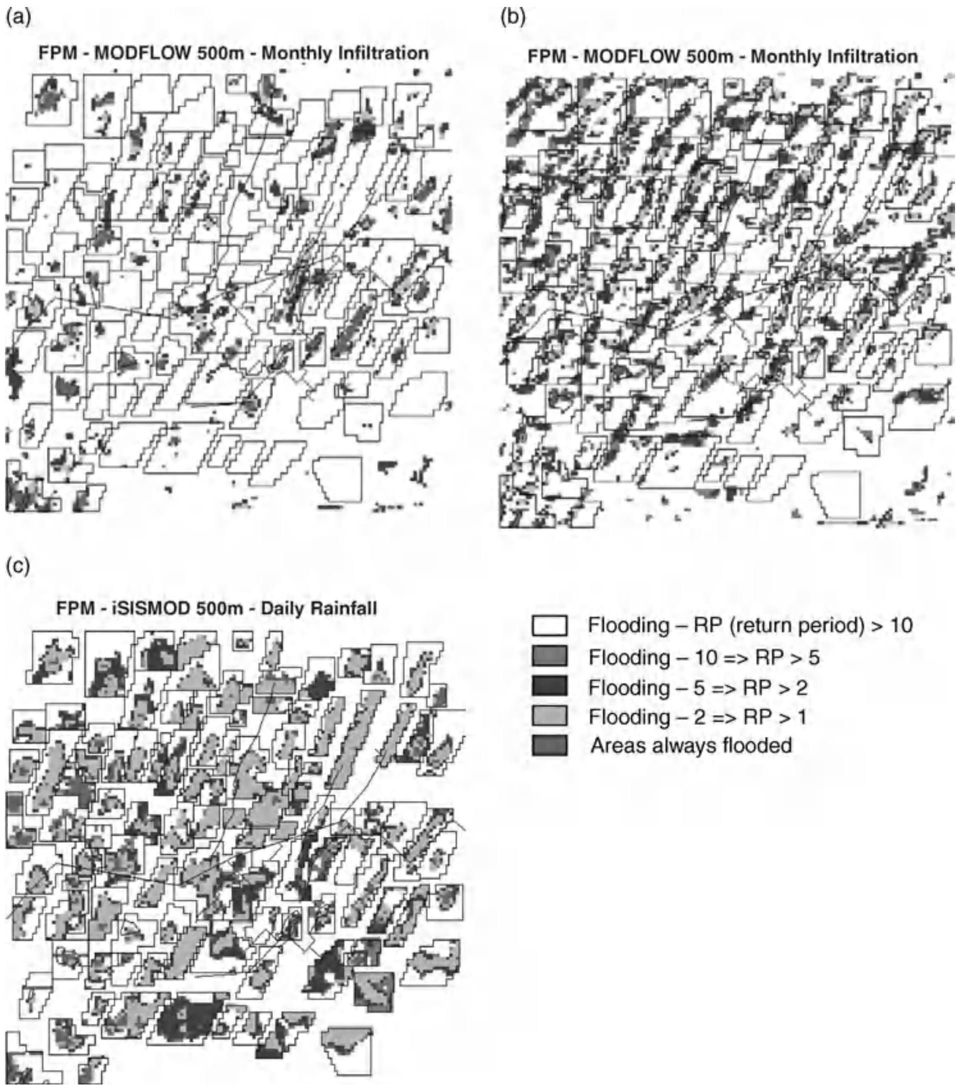


Fig. 22.12 Flood Probability Maps (FPMs). (a) MODFLOW – 5000 m – Monthly infiltration. (b) MODFLOW – 500 m – Monthly infiltration. (c) iSISMOD – 500 m – Daily rainfall. (see the colour version of this figure in colour plate section)

A Guide to Modelling Flooding in Large Lowland Basins

The final section of this chapter presents guidance on modelling flooding in large, lowland basins based on integrating the concepts and practical experience presented thus far. This includes development of conceptual flood risk models, synthesis of a multi-scale modelling framework to support integrated flood risk management, and utilization of the results obtained to support improved decision-making at all stages in the design of flood risk management strategies.

The key steps recommended when investigating flood risk in large, lowland basins are set out below and illustrated in Figure 22.13:

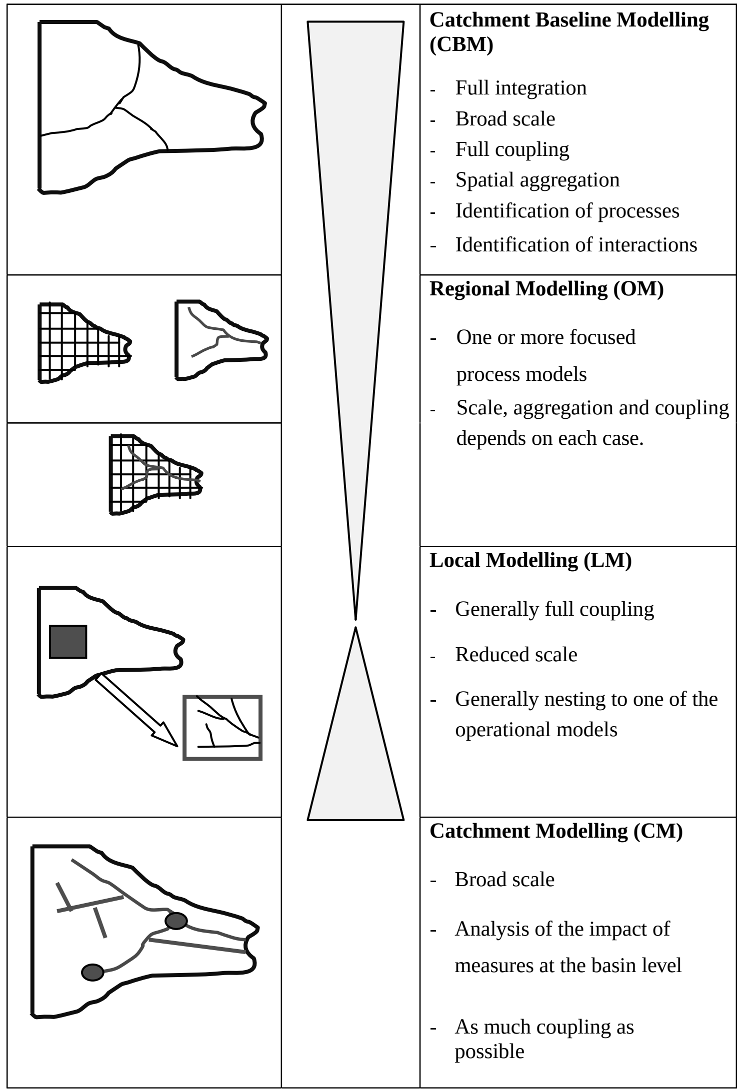


Fig. 22.13 Schematic representation of application of the Framework for Catchment Modelling Studies (FCMS) to catchment flood risk management studies.

Model conceptualization

Establish the relationships between the project objectives, the disciplines involved and the modelling requirements necessary to support decision-making at all stages of the project.

Identification of catchment theme and modelling theme for the project

This involves identifying and summarizing the issues that will drive and condition development of the modelling strategy. For example, in the case of the Río Salado Basin, groundwater-induced surface flooding was identified as the dominant flooding mechanism in the most economically important part of the basin (the Northwest – Region A) and could, therefore, be defined as the **catchment theme**. It followed that simulation of GW–SW interaction emerged as the dominant **modelling theme**, as explained in the previous section.

Baseline scoping model

The Baseline Scoping Model identifies all the relevant variables and flooding mechanisms that operate at the catchment scale and defines the roles they play within the conceptual risk model (s) developed in the initial conceptualization. The analytical tools applied in this step constitute 'interpretive' or 'generic' rather than 'predictive' models (Anderson and Bates 2001). This is the case because these tools should not necessarily require calibration as they are used mainly to guide development of a conceptual framework for exploring the sensitivities of the flooding system to the uncertainties inherent in different future scenarios.

Examples of the types of analytical tools that might be applied include:
- lumped rainfall-runoff models;
- simple tank models applied at the sub-regional scale;
- coarsely gridded, 3-D groundwater models;
- GIS-based regression models linking inputs (hydrometeorological data) with outputs (groundwater head responses).

A possible outcome of this stage could be a comparative matrix suitable for representing knowledge acquired concerning interactions between the dominant hydrological processes in the basin. This can then be used to determine which interactions are sufficiently strong to merit coupled modelling. An example is presented in Table 22.1. Once linkages between hydrological processes have been identified, it should also be determined whether they operate at the regional, sub-regional or local scales.

Table 22.1 Baseline Scoping Modelling – *Comparison* matrix

	Rainfall-runoff	Fluvial flooding	Ground water-induced surface flooding
Rainfall-runoff		3	3
Fluvial flooding	1		1
Groundwater-induced surface flooding	3	2	

Key:
1 Process A has **low** impact on Process B and **no coupling** is required.
2 Process A has **medium** impact on Process B and a **weak coupling** is required.
3 Process A has a **high** impact on Process B and a **strong coupling** is required.

Regional modeling

Regional modelling is performed to simulate the main flooding mechanisms identified in the Baseline Scoping Modelling step at a broad scale. Such operational models (which may be applied at the catchment or subcatchment scales) concentrate on simulation of the relevant components of the terrestrial hydrological cycle, with variable degrees of coupling, as indicated by the matrix developed in Step 3 (Table 22.1).

Local modelling

Local modelling is performed to explore and elucidate coupling between hydrological processes that are not simulated in the regional models. This step has two specific objectives: first, to gain deeper insights into the operation of key hydrological processes that can be fed back into one or more of the regional models (see feedback arrow in Fig. 22.9); and, second, to evaluate options for flood risk management, with the findings passed on to the final, Catchment Flood Risk Modelling step.

Catchment flood risk modelling

The final step is Catchment Flood Risk Modelling, which uses the outcomes of regional and local flood probability models (together with information on flood receptors) to produce catchment-wide maps of flood risk under current and 'with proposed flood risk reduction measure' conditions. These results are required at the project planning phase with the objective of analysing the flood risk benefits, costs and environmental impacts of proposed options for flood risk reduction at the basin scale, based on knowledge gained through local modelling.

Finally, Figure 22.13 summarizes schematically the above stages of the FCMS for the application into flood management studies.

Conclusions

The desire to develop and extend the representation of physical processes in modelling emerges from modelling campaigns or literature reviews as a natural human aspiration to exploit our constantly increasing knowledge of hydrological processes to the greatest degree possible when striving to produce scientifically credible flood risk management plans. However, in practice this aspiration is necessarily constrained by considerations of cost, time and data availability. The work reported in this chapter has attempted to balance the desire to model inherently complex groundwater and surface water interactions in large lowland areas explicitly with the need to work within strict budgets for time and effort through developing a modelling framework based on practical realism, a challenge that is often encountered when performing internationally funded projects in less economically developed countries (LEDCs). This challenge is particularly severe in large, low-lying basins where, besides the need to meet the objectives of multiple stakeholders, there also exists the need to overcome difficulties inherent in modelling a variety of flooding mechanisms and their interactions at a variety of scales.

It was recognized early in the research that no structured approach currently existed within which to perform the suite of modelling studies necessary to support integrated catchment flood risk management planning. The work then centred on an attempt to fill this void by developing a practically oriented Framework for Catchment Modelling Studies (FCMS) that fulfilled the need identified by Aradas (2001) for 'a staged and systematic approach to be used as a template for the development of modelling exercises to suit the physical characteristics of a basin and the level of detail required at each stage of a project, trying to strike a balance between project needs, cost and human resources.' The resulting FCMS parallels existing approaches for system-wide geomorphic studies, environmental assessments and engineering project management. Indeed, one of the lessons learned from the research studies is that geomorphic, environmental and engineering investigations must be closely linked to, and used to inform, the mathematical models used to generate maps of flood probability.

The applicability and utility of the proposed FCMS was tested in an Integrated Study of flood risk management options for the Río Salado Basin, with the modelling theme selected to focus on the flooding mechanism that was critical to flood risk management in this large, lowland catchment: groundwater and surface water interaction. It was found necessary to model flooding at both regional and local scales in order to address the variety of stakeholder issues, and a coupled modelling approach was found to be essential to representing the dynamics of GW–SW interaction at the local scale. The resulting Flood Probability Maps (FPMs) proved effective in synthesizing model outputs in a form suitable to support options appraisal and identification of sustainable flood risk management strategies.

The Río Salado studies revealed that, at a broad scale, regional modelling must focus on simulating only the dominant flooding mechanism (in this case groundwater-induced flooding) in order that modelling requirements and run times are manageable. However, it was demonstrated that using MODFLOW alone to generate FPMs could generate only a first approximation of the extent of flooding and was incapable of correctly simulating the depth of water on the surface.

The research further revealed that in low-lying areas groundwater heads predicted by MODFLOW are strongly influenced by evapotranspiration from the saturated zone and, to a lesser degree, the elevation of the ground surface. This is the case because the fact that horizontal fluxes are negligible makes the modelled groundwater profile more sensitive to changes in the input hydrological stresses and less sensitive to trends in the elevation of the ground surface.

Local modelling in a test area of the Northwest Region, using a fully coupled model (iSISMOD), demonstrated the justification for expending the additional resources necessary to model the delicate interplay between hydrometeorological stresses, groundwater heads and geomorphic features that is responsible for extensive and prolonged flooding associated with GW–SW interaction in a large, lowland basin.

The overall conclusion to be drawn from application of the proposed FCMS is that the modelling strategies it supports are capable of integrating key conceptual, technical and modelling activities in a way that supports systematic screening of issues during the early stages of a project, design of an appropriate modelling strategy during the middle stages of a project and, hence, efficient and justifiable deployment of resources to support the development of an integrated catchment flood risk management plan at the conclusion of a project.

Acknowledgements

This work is largely based on the outcomes of R.D. Aradas's PhD studies, which were performed under the supervision of C.R. Thorne and N.G. Wright, and which benefitted from intensive, multidisciplinary experience gained during execution of Master Plan studies for the Río Salado Basin. In view of this, thanks are due to all the team members who, together with the Unidad de Proyecto del Río Salado (Ministry of Public Works of Buenos Aires) gave generously of their insights and expertise to the worthy cause of reducing the adverse social and economic impacts of floods and droughts in the Río Salado Basin.

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23 Flood Modelling in the Thames Estuary

JON WICKS, LUKE LOVELL AND OWEN TARRANT

Introduction

Flood modelling is now an essential tool in the analysis of many of the components of flood risk management introduced in previous chapters. However, it is not always easy to decide on the most appropriate flood modelling method for specific tasks. This chapter uses case studies from the Thames Estuary to illustrate which flood modelling methods have been found to be most appropriate for a range of flood risk management analysis needs.

The chapter starts by providing background on the Thames Estuary and London covering historical flooding, flooding mechanisms, current flood risk management measures and a short history of Thames Estuary modelling. There then follow sections that describe how four flood risk management needs have been addressed using different types of modelling:

- One-dimensional (1D) hydrodynamic modelling of the main channel to generate extreme water level values for assessing defence levels and providing boundary values for use in local breach analysis (e.g. for flood risk assessments).
- Detailed 2D inundation modelling of the floodplain to produce flood maps for development control, awareness raising and other uses.
- Linked real-time 1D hydrodynamic modelling with local detailed 2D inundation modelling for flood forecasting, warning and other operational uses.
- Broad-scale 2D inundation modelling for use in appraising strategic flood risk management options (for the TE2100 project).

The chapter concludes with a discussion, based on the experiences of modelling the Thames Estuary, of where current modelling practice meets the current needs and where further advances are required.

Flooding and the Thames Estuary

Flood history on the Thames estuary

As a result of sea level rise, loss of marshland and general encroachment, the tidal limit of the Thames has slowly migrated upriver over the centuries. The high water level through central London has steadily increased, along with the potential for higher extreme water levels (Gilbert and Horner 1984). Notable floods occurred in 1099, 1236, 1663, 1791 and, more recently, in 1928, when 14 people were drowned. Most recently, a tidal storm surge in 1953 exceeded the 1928 level at London Bridge by 23 cm. Considerable damage and flooding occurred along the entire east coast of England and in total 300 people were drowned, many of them residents of Canvey Island on the Thames Estuary. Until this event, the 1928 level was the highest on record.

In 1965 a surge tide almost as high as that recorded in 1953 entered the Thames Estuary. Fortunately, this time there was little flooding as

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the flood banks downriver had been raised following the 1953 disaster. However, through central London where the flood defences had not been raised, water was reported to have lapped at the top of the river walls, apparently providing a graphic demonstration to Members of Parliament and perhaps ultimately influencing the eventual decision to construct the Thames Barrier and associated tidal flood defences.

Flooding mechanisms

On the Thames the tidal influence extends upstream from Southend diminishing towards the normal tidal limit at Teddington (Fig. 23.1). Upstream of the normal tidal limit flooding is usually a result of high fluvial flows from the upstream catchment, whereas downstream of the Thames Barrier, flooding from the Thames would be caused by storm surges and high tides. Between Teddington and the Thames Barrier flooding can be caused by a combination of tidal and fluvial sources, although storm surges provide the conditions for the most extreme water levels along most of the Estuary.

The Thames Barrier has an important influence on the hydraulics of the Thames Estuary. Upstream of the Thames Barrier, closing the structure soon after low tide creates a large 'reservoir' upstream. This enables fluvial flows to be stored, whilst preventing the high tides from flowing upstream into London. Downstream of the Thames Barrier, closing the Barrier can also influence water levels: a reflective wave may be set up, which can increase downstream water levels by several centimetres.

Thames tidal defences

The tidal Thames has an extensive, and in some respects, unique network of flood defences. The central feature of these defences is the Thames Barrier (Fig. 23.2). The Barrier is closed to prevent high tides flowing upstream, protecting central London from tidal flooding. The Barrier has been designed and is maintained to provide at least a 1000-year standard of protection until the year 2030.

Downstream of the Thames Barrier, continuous raised defences exist on both riverbanks. For some distance downstream of the Thames Barrier the crest levels of these defences are defined in statute and also ensure a current standard of protection of 1000 years. Further downriver, towards Southend, the standard of protection is more variable, with only the developed areas such as Canvey Island benefiting from such a high standard of protection.

Also downstream of the Thames Barrier there are a number of active barriers on tributaries, for example the Barking Barrier on the River Roding (Fig. 23.2) and the Dartford Barrier on the River Darent.

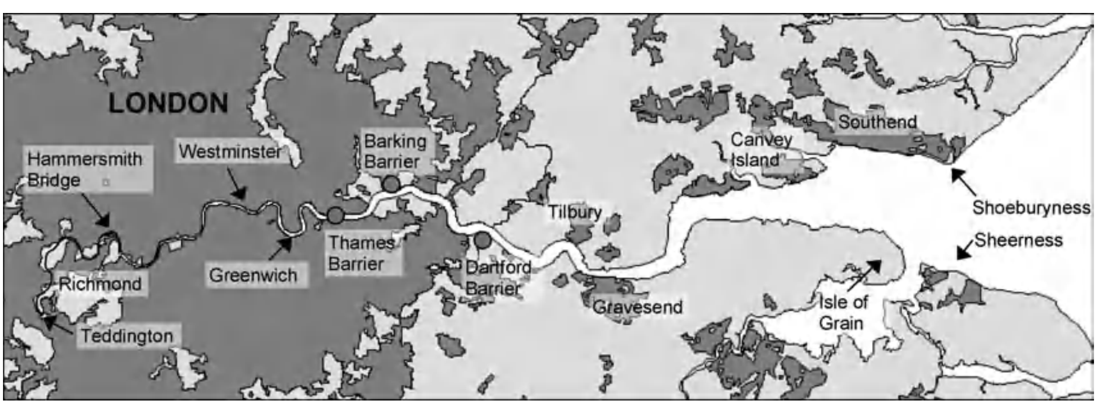

Fig. 23.1 Thames Estuary location plan. (See the colour version of the figure in Colour Plate section.)

Fig. 23.2 The Thames Barrier (left) and Barking Barrier (right).

Upstream of the Thames Barrier, formal defences exist in the form of raised walls, embankments and demountable defences. The upstream defences protect the area against normal high tides, perhaps combined with reasonably high (but not extreme) fluvial flow, ensuring that, for the time being at least, the Barrier should only have to close on a limited number of high tides each year.

The Thames Barrier is only closed when the forecast tide, in combination with the Thames fluvial flow at Kingston, is likely to produce water levels at London Bridge that are within 0.4 m of the defence crest level at this point.

Overview of modelling on the Thames estuary

Over the years several different hydraulic models of the Thames Estuary have been developed, both physical and mathematical. A number of organizations have contributed to developing and refining these hydraulic models and they have been extensively used on a variety of projects including:

- Design of the Thames Barrier, which used both early mathematical modelling, as well as physical scale models, constructed at the Hydraulic Research Station in Wallingford (now HR Wallingford).
- Assessment of extreme water levels, using an in-bank one-dimensional computer model of the Estuary and a detailed statistical analysis of fluvial flows and surge tides (Halcrow and CEH Wallingford).
- Production of the Environment Agency's Flood Map and 'Areas Benefiting from Defences', using a 2D mathematical model of the floodplain (Halcrow).
- Flood Forecasting, using a 1D model to predict high water levels along the entire Thames Estuary, and a 2D floodplain model to analyse the impact of breaching or overtopping (Halcrow and HR Wallingford).
- Analysis and appraisal of future flood risk management options using both 1D and 2D models as part of the TE2100 project (Halcrow and HR Wallingford).

The last four of these projects are described in the following sections.

Extreme Water Levels

Introduction

In 2005 and 2006 the Environment Agency commissioned Halcrow to undertake two studies to reassess design water levels, for a series of annual probability (or return period) events, along the Thames Estuary. Levels were produced for a number of sites between Teddington and Southend. The following sections provide an overview of the methodology used to calculate the levels and highlight some of the limitations inherent with estimated extreme water levels.

Calculation of design water levels

The method used to estimate joint probability water levels for different return periods is complicated and involves several steps. The approach has been reported more fully in Halcrow (2005) and in previous reports (e.g. Halcrow 1988). In essence the calculation involves two main stages; firstly using the 1D ISIS model (www.halcrow.com/isis) to estimate a matrix of water levels at each of the selected locations along the Estuary (known as structure functions), and secondly using a statistical model to calculate the design water level at each of those points, for a selection of return periods.

Calculating structure functions

Structure functions report a water level at a point along the Estuary as a function of a given sea level and river flow, and are determined by hydrodynamic modelling, which takes many factors into account (e.g. channel geometry, Thames Barrier operation, tidal propagation, etc.). Structure functions have been calculated for several points along the Estuary. Design tides (plus surge) at Southend were modelled at 0.5-m intervals and ranged from 1.5 m AOD (Above Ordnance Datum) to 5.5 m AOD for present day, 1.8 m AOD to 5.8 m AOD for 2052, and 2.1 m AOD to 6.1 m AOD for 2102. Fluvial flows ranged from 2 m^3/s to 1200 m^3/s. The ranges in levels and flows were set to ensure that results were available for the full extent of boundary conditions generated from the extremes analysis with climate change uplifts (from Defra 2006). Two illustrative examples of structure functions are shown in Figures 23.3 and 23.4.

Calculating return period water levels

This complex process (developed by CEH Wallingford; Halcrow 1988), calculates level-frequency results for a number of return periods and locations along the Estuary. The method uses records of sea level at Southend and river flows at Kingston (upstream of Teddington), along with the structure functions as inputs to their statistical analysis.

The analysis uses the data to calculate the probability of a given Southend water level occurring with a given fluvial flow, considering also the likelihood of a Barrier closure for that event. These probability data are calculated for neap-to-neap cycles and then converted to an annual maximum form, so that level frequency diagrams can be drawn for each location and water levels for a particular return period can be reported. To provide an indication of potential climate change impacts on river flows, the statistical modelling

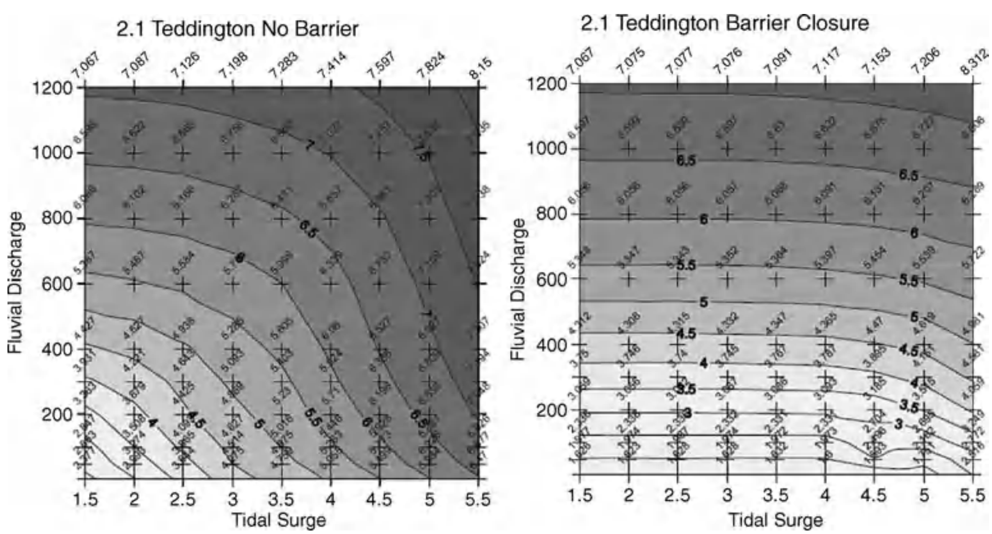

Fig. 23.3 Example structure function for Teddington (ISIS model node 2.1). (See the colour version of the figure in Colour Plate section.)

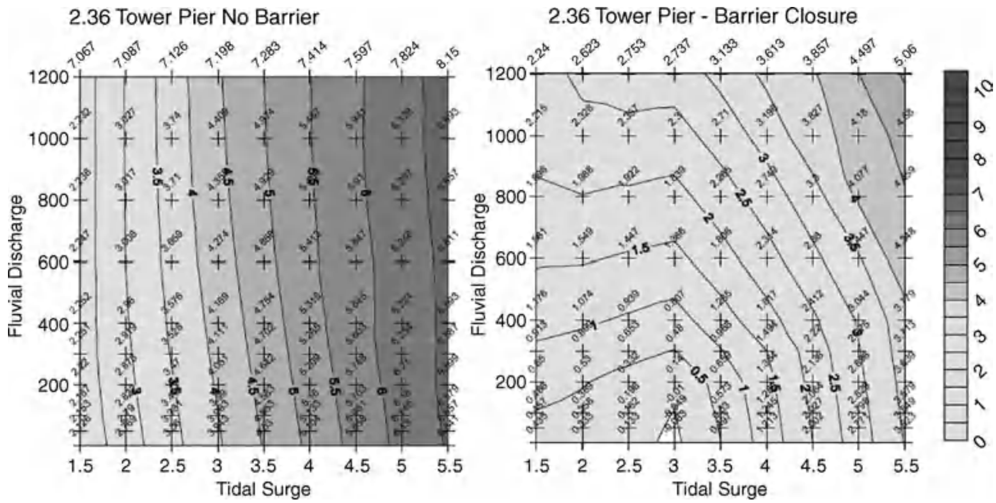

Fig. 23.4 Example structure function for Tower Pier (ISIS model node 2.36). (See the colour version of the figure in Colour Plate section.)

allowed for a 20% increase in Thames fluvial flow in 2052 and 2102 (the 20% increase was the current government advice from Defra, 2006).

Appropriateness of 1D modelling

The 1D in-bank model was selected as being most appropriate for this project due to a number of reasons including:

- The model was already in existence and had been calibrated to adequately represent in-channel water levels.
- The method of generating the structure functions required two sets each of 81 simulations to be undertaken for each climate scenario; the modelling approach had to be fast enough to ensure that some 500 simulations could be undertaken within a reasonable period of time. As the 1D model can simulate a few days' tides in less than 1 minute this criterion could be achieved.
- Modelling of the floodplain was not required as the project objective was to determine extreme water levels without flooding. The 'glass-walled' 1D model is a very suitable method to achieve these results (in a 'glass-walled' model both extremes of the simulated channel cross-section rise vertically to levels higher than any simulated water levels).
- Wind-generated wave effects were not required to be predicted – in the outer Estuary these can be significant and can be added as a post-process to the generated 'still water' levels.
- The operation rules for the Thames Barrier, Richmond weir and Teddington weir could be idealized and included in the models in a consistent fashion.

Flood Mapping

Background

The Environment Agency's Flood Map (www.environment-agency.gov.uk) is a set of maps aimed primarily at increasing awareness of flood risk. Flood Zone data form the base layers for the Flood Map and are used to provide information for planning consultation compliant with UK government planning policy statement PPS25 (which defines the process used in England to avoid inappropriate development in areas at risk of flooding). Flood Zones are geographical areas defined by

Table 23.1 Flood Zones definitions

Flood Zone	Annual probability of flooding
1	'Low probability': less than 1 in 1000 (<0.1%)
2	'Medium probability: between 1 in 1000 (0.1%) and 1 in 100 (1%) for river flooding, 1 in 200 (0.5%) for flooding from the sea
3a	'High probability': 1 in 100 or greater (>1%) for river flooding and 1 in 200 (0.5%) or greater for flooding from the sea

their annual exceedance probability of flooding (Table 23.1) and depict the extents of flooding if there were no flood defences. An associated map is the 'area benefiting from defences' – a map showing areas that would flood at the 1% annual exceedance probability (0.5% for coastal flooding) if formal defences were not in place/operational.

The Flood Zones and 'areas benefiting from defences' maps for the Thames Estuary between Teddington and Dartford were generated using a combination of 1D and 2D modelling as described below (Halcrow 2006).

Modelling methodology: flood zones

To generate the Flood Zones for the Thames Estuary upriver of Dartford, hydraulically discrete areas of tidal floodplain, known as embayments, were modelled individually using a 2D floodplain-only model (developed using the TUFLOW software; www.tuflow.com).

Inflow boundaries were derived for each embayment model using the 1D ISIS model of the Thames Estuary, which used a tidal surge event with peak water levels equivalent to the extreme water levels calculated in the work described above (see 'Extreme water levels'). These boundaries were then used to run the 2D 10-m grid models. The numerical grids used in the 2D modelling were constructed using a digital terrain model (DTM) derived from LiDAR (light detection and ranging) data. Filtered LiDAR was used for the floodplain modelling with a 10-m computational grid. Buildings in the floodplain were represented through the use of higher roughness values. The use of a smaller grid size would have resulted in unacceptably long run times. Consideration was given to other ways of representing buildings (such as using unfiltered LiDAR, imposing the buildings back onto the filtered data through use of MasterMap layers and using a 'porosity' function) but these were discounted as either impractical or unproved/inaccurate following initial trials. Flood defences had to be removed from these data prior to the construction of the model as Flood Zones, by definition, show the undefended floodplain for various annual probabilities of flooding. Flow routes were checked and schematized using aerial photography.

The outputs from the 2D models were then post-processed to remove small 'dry islands' (potentially erroneous dry areas surrounded by flooding, caused, e.g., by remnant man-made features commonly present in LiDAR-derived DTMs).

Modelling methodology: areas benefiting from defences

The Thames Estuary between Teddington and Dartford benefits from defences that currently provide protection against events with a greater than 0.1% probability of occurrence. Such a high standard of protection is probably an exception nationally, and the Environment Agency has taken the decision to show the area benefiting from defences in London going out to the extreme flood outline (i.e. the 1000-year event) rather than the 200-year event normally used for tidal flooding.

The same approach to modelling was used as for the Flood Zones but with the flood defences added to the model. The area benefiting from defences was calculated as the difference between the with-defences and no-defences flood extents.

Appropriateness of modelling methods

For the flood mapping, non-linked 1D (channel) and 2D (floodplain) models were considered most appropriate, with each embayment modelled separately. This approach intentionally maximized the simulated flood extent (there was no feedback from the flow entering the floodplain to act to reduce inflow from the river). Alternative approaches using fully linked models were considered but were rejected as they would not have achieved the Flood Map specification and may have led to underestimates of risk.

A sensitivity analysis was undertaken using variations in ground data (LiDAR, synthetic aperture radar (SAR)), roughness coefficient (base, $n = 0.050$), boundary water level (base, base $+0.3$ m, base -0.3 m) and model grid size (5 m, 10 m, 20 m). The analysis suggested that results are most sensitive to ground data and model grid size. For defining Flood Zones, a grid size of 10 m derived from LiDAR was found to be optimum in terms of balancing run times and resolution of results.

Flood Forecasting

Introduction

Flood forecasting models need to run quickly and robustly to provide sufficiently accurate predictions of water levels at key 'action' thresholds up to 36 hours ahead. The Environment Agency's flood forecasting system for the Thames Estuary consists of a Proudman Oceanographic Laboratory 2D model of the North Sea and eastern English Channel linked at Southend to an ISIS 1D model of the Estuary. The 1D forecasting model runs both as part of the Agency's National Flood Forecasting System and as part of Themis. Themis is a software system developed for the Agency for real-time flood simulation including the production of real-time flood maps (Fig. 23.5).

The Themis system

The Themis software was developed to automatically link with existing Agency systems to receive the latest predictions of Teddington fluvial flows and Southend tides/surges. The software allows users to manually adjust boundary data and add potential/actual defence breach locations before undertaking 2D and/or 1D simulations. Operation of the Thames Barrier can also be

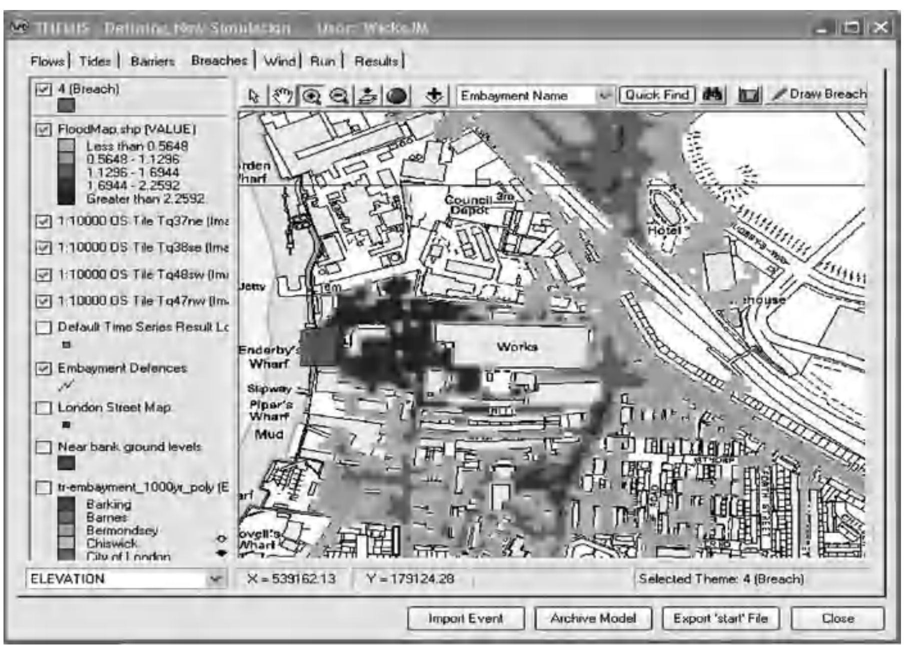

Fig. 23.5 Predicted flooding following a hypothetical breach at Enderby's Wharf. (See the colour version of the figure in colour plate section.)

included in the simulation, and 'what if' scenarios undertaken to help optimize operational decisions. Simulation results are automatically processed to generate tabulated maximum water levels, time series plots of water levels and flood extent maps, which can all be viewed in Themis or exported to other systems. The software is populated with 2D models covering the 23 Thames Region embayments, 1D flood cell models providing simplified but quick-running models of the 23 Thames Region embayments and 12 Anglian and Southern Region embayments, and the 1D Tidal Thames model. The 2D and 1D flood cell models are dynamically linked to the 1D model. Further details of the system are provided in Tarrant *et al*. (2005), with further information on the selection of models provided in Wicks *et al*. (2004).

Appropriateness of modelling methods

For forecasting, the run times and robustness of models are of crucial importance, and lower accuracy may well be acceptable in order to achieve fast run times. The objective is to provide reliable evidence to assist decision-making in areas such as issuing of warnings or operation of barriers. Use of a detailed model to achieve an accuracy of 25 mm but requiring 1 hour to run is not appropriate if a simpler model can achieve 100 mm accuracy in a 1-minute simulation. A range of models are provided for flood forecasting in the Thames Estuary so that simple and quick models can be run rapidly first, and then more complex (and slower) models can be run later to provide refined results (Fig. 23.6). For example, simple pseudo-2D flood cell models can be run initially to assist in immediate decision-making, followed by the slower 2D TUFLOW models to provide refined outputs.

TE2100: Strategic Flood Risk Management

Introduction

The Environment Agency's Thames Estuary 2100 project (TE2100) was set up to develop a plan for

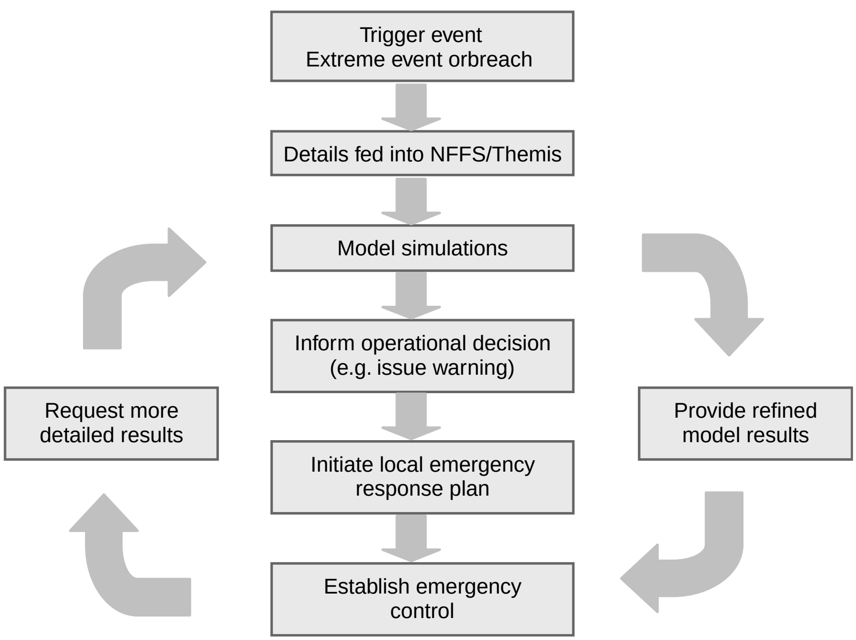

Fig. 23.6 Modelling to support operation decision-making.

flood risk management in the Thames Estuary for the next 100 years. The production of the plan was supported by a programme of modelling designed to improve understanding of the baseline behaviour of the system and predict future behaviours both with and without changes in flood risk management. Hydraulic modelling included both 1D and 2D modelling: the Estuary was modelled in both 2D (Telemac-2D) and 1D (ISIS), with the floodplain modelled using 1D flood cells (ISIS) and 2D (TUFLOW). In this section two examples of the TE2100 flood modelling are described:

- 1D modelling to help understand the 'limits of engineering adaption';
- linked 1D and 2D modelling to help appraise options.

Limits of engineering Adaptation

The 'Limits of engineering adaptation' study was initiated by the TE2100 team to gain an early appreciation of the likely limits of large-scale 'hard engineering-biased' flood risk management options against sea level rise and future increases in storm surge events. One of the main aims of the study was to define the various points, in terms of sea level rise, at which engineering responses would face a critical threshold that would force a further adaptive change in the system. The study thus defined a number of key adaptation thresholds, as follows:

- Threshold 1: the point at which the freeboard allowance within the existing flood defences is eroded by a given surge event, or by a future spring tide.
- Threshold 2: the point at which the height of existing downriver defences and the crest level of the existing Thames Barrier would need to be raised.
- Threshold 3: the point at which the existing Thames Barrier and associated walls and embankments cannot be adapted further, leading to the possible move to an outer estuary barrier (e.g. at Southend).
- Threshold 4: the point at which it is necessary to modify the structure at Southend into a barrage.
- Threshold 5: the point at which it is considered impractical to intervene further to manage flood risk through engineering (i.e. the overall engineering limit to adaptation).

The approach taken for the study was to use extensive 1D hydrodynamic modelling together with application of engineering judgement. Each engineering response was tested against a range of sea level rise scenarios ranging up to a maximum of 8 m above the current mean sea level. The engineering responses explored were constrained to: (i) raising of defence walls and embankments; (ii) modifying the Thames Barrier; and (iii) construction of new throttles, barriers and barrages. ISIS models representing different future flood risk management responses were run with a series of extreme tides at Southend to calculate in-bank water levels at a number of locations along the Thames. These water levels were then compared with defence data (i.e. information on current and future crest levels), with an allowance for freeboard.

The 1D models were schematized as in-bank, meaning water could not spill out onto the floodplain. Thus the results for each scenario could be reused to define overtopping thresholds for different defence levels. Once the simulations were completed, the maximum water levels from each design event were extracted and stored in a spreadsheet, which was then used to facilitate analysis. Various overtopping thresholds were then calculated for each response, both with defences at current levels, and with defences raised by 1 m.

A range of assumptions and simplifications were needed to ensure the study remained tractable; these included: ignoring increases in fluvial flows, using a representative surge shape, simplifying the barrier closure rules and using an in-bank 1D model.

The study indicated that, assuming that the requirement is to maintain a 1:1000-year standard of protection for the urbanized embayments along the Thames Estuary, the absolute maximum rise in mean sea level that the potential engineering adaptations tested in this study could accommodate is:

- 5.25 m (for a 1-m increase in surge magnitude); or

- 5.75 m (with a 0.4 m increase in surge magnitude); or
- 6.0 m (with no increase in surge magnitude).

Fortunately, a sea level rise of 5.25 m by 2100 is higher than that postulated for any of the climate change scenarios currently in use by the TE2100 project (e.g. the 'High + +' scenario allows for 3.2 m of mean sea level rise combined with 1 m increase in 1:1000 year surge by 2100).

It should be noted that this work did not explore the economic, social and environmental 'costs' associated with the implementation of the responses. The 'of some of the responses explored may indeed be prohibitive, thus narrowing the envelope of sea level rise that could be adapted to with engineering or structural responses alone. A portfolio of responses – both structural and non-structural – will have to be employed in the Thames Estuary in order to adapt to future flood risk.

The use of a 1D hydrodynamic model was the appropriate method to enable many hundreds of simulations to be undertaken within a short project programme. As flood risk (flooding probability × consequences) was not required to be calculated the 'glass-walled' 1D model did not need to extend onto the floodplain. It should be noted, however, that in this high-level study the application of engineering judgement was an equally important element as the modelling.

Appraisal of options

The TE2100 project included the formal appraisal of a set of strategic flood risk management options for the Thames Estuary. The appraisal required a range of flood risk metrics to be calculated including direct property flood damages, 'risk to life' estimates and a large set of floodplain depth-probability grids (Wicks *et al*. 2009). The depth-probability grids are used in subsequent analysis to inform wider indicators of social, economic and environmental impact. The main components of the method used for the modelling were:

- Fully hydrodynamic ISIS 1D modelling of the tidal Thames for a range of tidal and fluvial events.
- Fully hydrodynamic but relatively low-resolution broad-scale 2D modelling of floodplain flow (dynamically linked to the 1D model of the river).
- Treatment of breaching through a combination of breach factors and embayment-scale TUFLOW 2D breach models.
- Simulation of deterioration of defences over time.
- Explicit consideration of the main source of uncertainty (Southend extreme water level).
- Likelihood of defence failure included as a function of defence type and water level in the river (represented by fragility curves).
- Inclusion of the additional risk due to potential failure to operate the Thames Barrier (and proposed barriers).
- Calculation of flood depth probabilities resulting from defence breaching, overtopping and barrier failure.
- Direct property damages (including residual risk) calculated to derive annual average damage (AAD) and present value (PV) damages.
- Estimation of annual risk to life based on the method described in Defra (2008).

Two baselines ('walk away' and 'maintain existing') and seven strategic option sets were simulated for a range of epochs up to 2170 for two climate change scenarios.

The use of broad-scale 2D modelling of the floodplain was selected as appropriate at it was essential to estimate floodplain flood depths – the use of detailed 2D modelling would have resulted in unacceptable run times given the project programme (many hundreds of simulations were required). The use of non-fully hydrodynamic modelling of the floodplain was also considered but rejected as it was not considered sufficiently accurate for appraisal.

Conclusions

As described above, a range of flood modelling methods has been found to be necessary to meet the needs of the Environment Agency on the Thames Estuary:

• Calculation of extreme (in-channel) water levels required the use of a 1D model.
• Production of flood maps required detailed 2D modelling of the floodplain with boundary conditions provided by a non-linked 1D model of the channel.
• For flood forecasting a 2D model of the sea and outer Estuary is linked to a 1D model of the Estuary upriver of Southend. Where the potential area impacted following a breach is required to be forecast, then a detailed 2D model of the floodplain is provided as part of the Themis inundation modelling system.
• For analysis of strategic flood risk management options a 1D model has been used to assess threshold of adaptation. For appraisal of options a dynamically linked 1D (channel) and broad-scale 2D (floodplain) has been used to enable property damages and other impacts to be estimated.

The selection of the most appropriate modelling methods is not necessarily easy and requires experienced modellers to consider a range of criteria, including: flow mechanisms, required accuracy, availability of existing models, time available for modelling, outputs required and availability of suitably skilled modellers. Even when the most appropriate method is used, there will remain key assumptions to make and uncertainties to understand and communicate to end users. Some key uncertainties that apply to much of the modelling described in this chapter are discussed below.

Whilst river water levels can be predicted with some confidence (for given boundary conditions), the prediction of maximum flood extents, floodplain velocities and floodplain water depths has much lower confidence (both because the processes are harder to simulate and because there is a lack of observed data to confirm accuracy). In addition, there are many parameters and phenomena that we include in our models over which we need to make informed judgements or limiting assumptions in order to include, for example:
• how or when a defence might fail;
• the forecast shape and coincident timing of a surge with a high astronomical tide level;
• the probability of a particular flow occurring with any particular tide;
• future extreme water levels (sea level rise and increased surge size).

There are many parameters and phenomena that we do not include in our hydraulic models (e.g. due to lack of data, lack of mathematical representations or lack of deterministic knowledge), including:
• non-standard human behaviour effects on operation of river structures;
• river bed sedimentation/erosion interactions with hydraulics;
• unpredictable weather effects;
• extremes or failure that we have not experienced or have been unable to conceptualize.

Acknowledgements

This chapter references material produced by Halcrow for the Environment Agency. The latter's help and encouragement is gratefully acknowledged although the views and opinions expressed do not necessarily reflect those of the Environment Agency. The most up-to-date data on Thames Estuary flood risk should be obtained directly from the Environment Agency.

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24 A Strategic View of Land Management Planning in Bangladesh

AINUN NISHAT, BUSHRA NISHAT
AND MALIK FIDA ABDULLAH KHAN

Introduction

Situated in the lower reaches of the three great rivers, the Brahmaputra, Ganges and Meghna, Bangladesh is one massive alluvial floodplain criss-crossed by a network of several rivers, their numerous tributaries and canals. The system of rivers, canals, floodplains and water bodies is intensively integrated, influences the people's way of life and is fundamental to the country's mainly agrarian economy.

Given the hydrological setting of the country, for centuries water has been the lifeline of the country and at times has also been at the root of its sufferings. Floods inundate the landscape during the summer monsoon every year as rainfall and snowmelt from the mountains cause the rivers to spill over their banks. These inundations frequently develop into devastating floods. Land management in Bangladesh very much reflects the inundations, seasonal cycles of water availability and agricultural production. In keeping with the main theme of the book, this chapter discusses land management in Bangladesh in the context of flood management.

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Profile of Land Resources

The total area of Bangladesh is approximately 14.4 million hectares of which 12.46 million hectares are land surface and 0.94 million hectares are rivers and other inland water bodies. The actual areas fluctuate slightly due to changes taking place in the courses of major rivers creating new land through accretion in some places and devouring land through erosion elsewhere (Abdullah *et al*. 1991).

The extensive floodplain of the three major rivers and their tributaries and distributaries dominates the physiography of the country. About 80% of the country is floodplains composed of predominantly recent alluvial deposits transported by the rivers from the Greater Himalayan region. The entire country is low-lying and extremely flat with the exceptions of a few hills in the north, northeast and southeast of the country. Hill areas in the northeastern and eastern parts occupy about 12%, and terrace areas in the centre and northwest occupy about 8% of the country. Because of the flat topography flooding spreads evenly and accumulates on the plains. The alluvial rivers have natural levees on both banks, which slope down towards the floodplains. There are numerous natural depressions, mainly in the northeast part of the country, locally known as haors, in the northwest region known as beels, and in the southwest part of the country known as baors. Figure 24.1 shows the generalized physiographic features of the country.

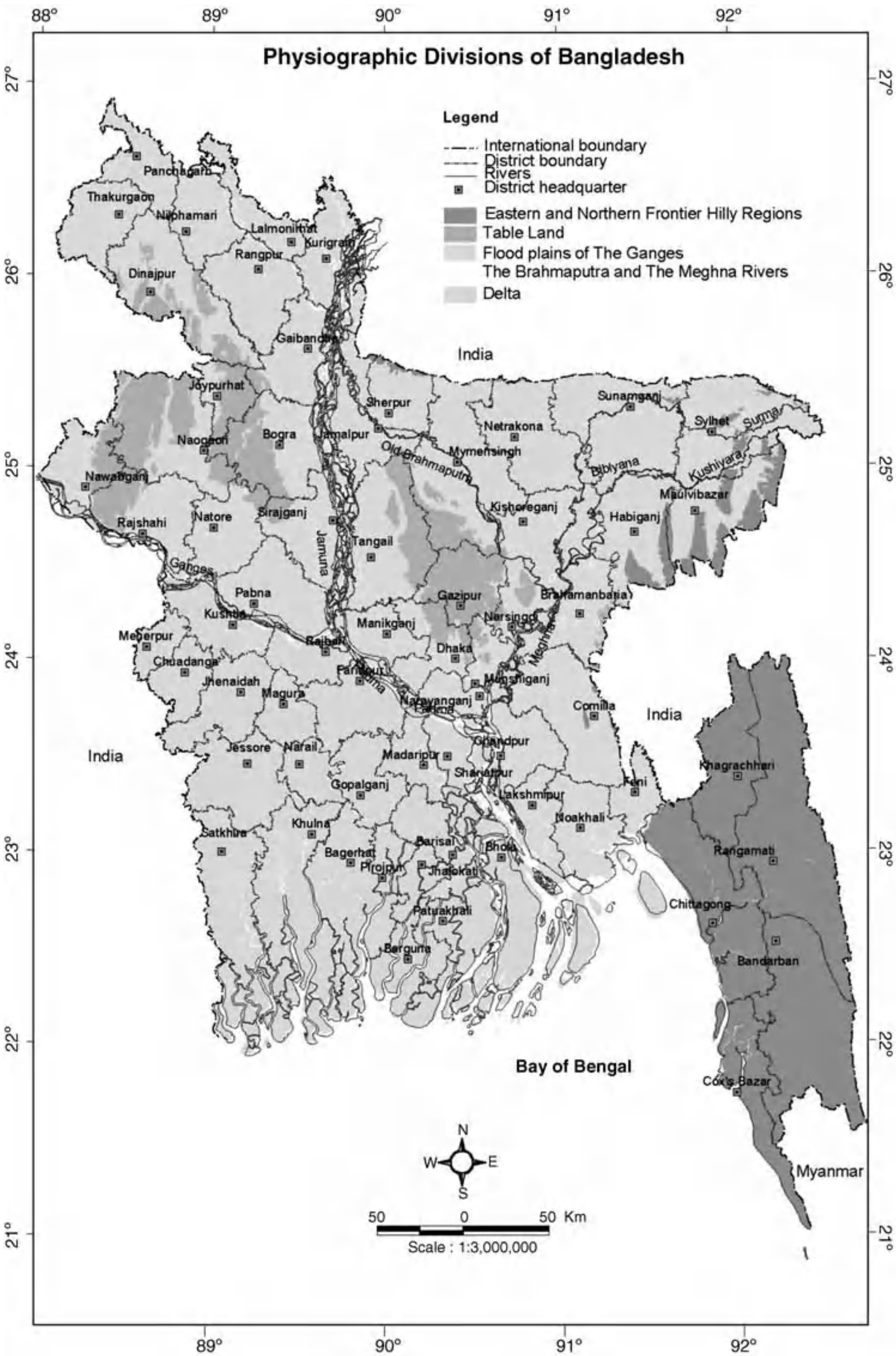

Fig. 24.1 Map of Bangladesh showing the physiographic features of the country. (See the colour version of this figure in Colour Plate section.)

By the end of the 1960s, a sort of equilibrium had been reached in the pattern of land use, which had remained unchanged over the years (Abdullah *et al*. 1991). Agriculture dominates both land use and the national economy. Arable land stands at about 8.9 million hectares, forest accounts for about 2.02 million hectares, and settlements plus water for about 3.03 million hectares. Floodplain settlements typically are concentrated on the highest available land, on river banks or ridges. They are surrounded by agricultural land extending down to the lowest land, which is deeply flooded in the monsoon season. Currently a total of 8.45 million hectares of arable land is cultivated. These figures indicate that land in Bangladesh is already very intensely used.

Moreover, with increased focus on industrialization and a shift of the population towards urbanized areas, the demand for non-agricultural use of land is increasing gradually, especially in and around the major cities. According to the Bangladesh Bureau of Statistics, 9567 km^2 of the country is urbanized (Bangladesh Bureau of Statistics 1991). This area includes urban growth centres, Thana (smallest administrative units) headquarters, towns, cities and statistically metropolitan areas.

Land Management Practices: Reconstruction of History

Since ancient times, land management in Bangladesh has centred on the numerous rivers, the annual cycles of flooding and the hydrological characteristics of the country. In his famous work 'Ain-I-Akbari', the historian and member of Emperor Akbar's court claimed that Bangala (ancient name for Bengal) is derived from the combination of *vanga* and *al*. *Vanga* means wetlands and *al* is used for small embankments. According to his writings, '...the original name of Bengal was Bang. Its former rulers raised mounds ten yards in height and twenty yards in width throughout the province which were called al. From this suffix, the name Bengal arose' (quoted in Majumdar, 1943). The interpretations of Abul Fazal's writings as quoted above have been subject to much debate. But the text indicates the presence of embankments in ancient Bengal, and this has so far not been contested. The rationale for ancient rulers building such earthen structures, as the literature suggests, was flood protection and protection of agricultural production. Artificial cuts or breaches towards the end of the monsoon ensured drainage and natural fertilization of crop fields. These cuts were closed at the beginning of the monsoon. In many cases the embankments were built for philanthropic reasons, but also the embankments ensured the state's own interest, as agriculture was the only source of income in ancient times.

Some of these initiatives remain as relics of a bygone era. For example, a large embankment in the Natore district containing masonry bridges and big enough to carry a railway bridge was constructed by Rani Bhawani (1716–1795) and is called Rani Bhawani's Jangal.

During the Moghul period (1526–1757), government institutions such as the Pulbandy or Pushtbandy were created at regional level and allocated government funds to carry out activities to ensure construction and maintenance of roads, embankments and dredging of rivers. Locally, it was mandatory for *zamindars*, or landlords, to initiate and supervise management of local land and water resources. If needed, zamindars were authorized to levy taxes known as *Abwab* and *Mahut* taxes, to compensate for any fund deficiencies. The land- and water-related issues of a village were the responsibility of the *gram-sharanjami*, comprising volunteers, who were controlled by the *Panchayat*, a community group consisting of village elders.

However, towards the end of the Moghul period these land management and flood protection works started to get disorganized and unregulated. During the Colonial Period (1757–1947), the water management taxes and *gram-sharanjami* did not exist anymore, and state support to the *Panchayat* was withdrawn. The colonial revenue systems in British India were based on land with no emphasis on water (Alcoli 1921). Under the changed circumstances the community was kept at bay and local land management institutions and existing practices were gradually eroded. The colonial administration tried to reduce flood damages by strengthening the existing embankments, and building new embankments without understanding the existing hydrology. To work, tenants had to breach the embankments but the authority, with little knowledge, tried to stop these breaches. Towards the end of the Colonial Period, with the depth of the rivers reduced due to siltation and the embankments deteriorating due to lack of maintenance, frequent flooding caused untold suffering for the people.

The modern period of public initiatives in flood management began in 1957, with the appointment of the Krug Mission in the wake of the devastating floods in 1954, 1955 and 1956. At the request of the then government of Pakistan, the United Nations sent this technical assistance mission, which emphasized the need for flood control to increase agricultural production. The East Pakistan Water and Power Development Authority (EPWAPDA), now known as Bangladesh Water Development Board (BWDB), was created in 1959 and was responsible for the planning, design, operation and management of all water development schemes. Consequently, a Master Plan for water development, formulated in 1964, included a portfolio of 58 land and water development projects for implementation over 20 years. On the basis of this plan, large-scale land and water development schemes were initiated by the government (Datta 1999).

In 1972, the Land and Water Sector Study changed the emphasis to agricultural production rather than flood control, and underlined the need to consider land and water as integrated resources. Recommendations included the development of minor irrigation through low-lift pumps and tube wells supported by complementary less capital- and labour-intensive Flood Control and Drainage (FCD) projects. The FCD schemes are formulated to achieve three principal and distinctive goals (MPO 1986):

- To minimize damage and destruction caused by catastrophic floods and storms.
- To provide safety for lives and property and to minimize damage and disruption of essential economic activities.
- To increase agricultural production through changes in crop type and cropping patterns.

It should be noted that FCD schemes are not only used for flood control and drainage, but also have the additional objectives as mentioned above. Moreover, irrigation through low-lift pumps (LLP), tube wells or traditional irrigation practices and devices are often components of the scheme. For this reason, the term 'FCD' scheme is often used interchangeably with 'Flood Control, Drainage and Irrigation' (FCDI) scheme (Ali 2002).

The National Water Plans (NWP) of 1987 and 1988 were initiated by the government to prepare a comprehensive water master plan with a planning horizon up to 1990–2010. The Master Plan Organisation, known as MPO, was created to address the different issues leading to water resources management and develop these plans. The National Water Plan stresses that:

> '...The largest and most significant impact of flooding is the limiting effect that average flood levels and the risk of inundation have on the choice of crops by farmers in the monsoon and dry seasons. Factors including the normal depth of inundation, the risk of flooding and poor drainage force farmers to choose low yielding crop varieties that can withstand the expected flood depth and to employ very low intensity cropping patterns'.

Despite the many limitations of the plans, both phases of the NWP made vital contributions to the knowledge and understanding of the water resources of Bangladesh. NWP data have provided the basis for much subsequent water and land planning. In order to evaluate the potential land in terms of the nature and depth of annual flooding, the MPO formulated a framework of flood depth distribution through a classification of land types according to flood depth. This is the first and most reliable national land-type database, used in analytical activity and national planning issues.

The Flood Policy Study formulated in 1988 and 1989 set 11 guiding principles for future flood management in Bangladesh. These principles have been incorporated in the National Water Policy (Box 24.1). After the disastrous floods of 1987 and

Box 24.1 Eleven guiding principles introduced by the Flood Policy Study (1989)

1 Phased implementation of a comprehensive Flood Plan aimed at:
- protecting rural infrastructure;
- controlled flooding to meet the needs of agriculture, fisheries, navigation, urban flushing
and annual recharge of surface water and groundwater resources.

2 Effective land and water management in protected and unprotected areas.
3 Measures to strengthen flood preparedness and disaster management.
4 Improvement of flood forecasting and early warning.
5 Safe conveyance of the large cross-border flows to the Bay of Bengal by channelling them through the major rivers with the help of embankments on both sides.
6 River training to protect embankments and urban centres.
7 Reduction of flood flows in the major rivers by diversion onto major distributaries and flood control relief channels.
8 Channel improvements and structures to ensure efficient drainage and promote conservation and regulation.
9 Floodplain zoning where feasible and appropriate.
10 Coordinated planning and construction of all rural roads, highways and railway embankments with provision for unimpeded drainage.
11 Encourage popular support by involving beneficiaries in the planning, design and operation of flood control and drainage works.

1988, a number of studies on the flood problem of Bangladesh were carried out under the Flood Action Plan (FAP). Based on the 11 guiding principles, FAP was to set the foundation for a long-term programme to achieve a more permanent and comprehensive solution to the flood problem and to create an environment for sustained economic growth and social improvement. FAP emphasized the need for substantial changes in flood management ideas and approach, and controlled flooding rather than no flooding was advocated. The concept of compartmentalization was introduced and is currently under test in the north-central region of Bangladesh. The objective of compartmentalization is to regulate floods within certain desirable ranges coinciding with local needs. According to the National Water Management Plan Project (NWMPP 2000) the main legacies of FAP are as follows:
- Increased emphasis now being placed on social and environmental aspects as well as fisheries, navigation and the need for full popular participation and consultation.The justification for large-scale public sector FCD and irrigation projects has come under increasingly stringent review.
- The FAP regional plans and the Meghna Estuary Study provide a useful basis for integrated water resource planning at the regional and subregional level and as inputs for national planning - despite changes in water sector policy and strategy since their formulation.
- Detailed Guidelines for Project Assessment (GPA), Environmental Impact Assessment (EIA) and Social Impact Assessment (SIA) were produced.
- Useful practical experience with specific technical options has been gained through the pilot projects, particularly on river bank protection, compartmentalization, flood platforms, fish passes and dredging.
- Substantial further improvements have been made to the country's database, particularly with spatial data, and a wealth of studies is available.

The rehabilitation and improvement of existing schemes, rather than new development, was given attention in FAP. The FAP Summary Report, published in December 1995, presented a proposed framework and short-term (1995–2000) programme for future development. This implementation programme centred on completion of FAP activities, floodproofing, river management and coastal protection, urban FCD, and water and flood management. The Bangladesh Water and Flood Management Strategy (BWFMS) was the major strategy follow-up to FAP and became the working policy document for the water sector. Many of the BWFMS concepts were carried forward into the National Water Policy, which was adopted in December 1998. The concepts advocated a multi-sectored integrated approach, which was a substantial deviation from the ideas that had dominated water sector thinking until the early days of FAP.

The National Water Management Plan (NWMP), formulated in 2004, is a framework plan to be implemented by 24 organizations. Line agencies and other organizations are expected to plan and implement their own activities in a coordinated manner following government rules and procedures and according to their mandates within the NWMP. The Plan takes into account lessons learned from past activities in flood and land management and provides a comprehensive focus on relevant issues. The Plan is to be a rolling plan to be reviewed and updated every 5 years, providing a firm plan for the first 5 years, an indicative plan for the subsequent 5 years, and a perspective plan for the long term (25 years).

Over the past three decades flood and water management in Bangladesh has gradually evolved to its current stage. Although the FCD/FCDI projects still dominate management practices, considerable change and progress based on practical experience (and 'lessons learnt'), can be seen in the planning and implementation of these projects. Emphasis is on a more comprehensive approach with multi-objective and multi-sectoral planning, rather than single-objective (e.g. food grain self-sufficiency) planning. The generally better performance of small-scale interventions and private sector development has seen the reduced importance of large-scale government projects. Stakeholder participation and consultation at all levels, from the private and NGO sectors as well as the public sector, is now recognized to be essential (NWMPP 2000). Low-cost activities such as structural floodproofing and non-structural measures are now advocated for rural areas, which are not of high economic importance, with urban FCD being given more emphasis than in the past. Present integrated water and land use planning is set to focus on the development of flood plain zoning to accommodate necessary engineering measures and allocate space for habitation patterns, economic activities and environmental resources (Flood Plan Coordination Organisation 1995).

Physical Basis of Land Management in Bangladesh

Land management in Bangladesh is determined mainly by the monsoon rainfall and the seasonal flooding that affects the greater part of the country. These physical determinants are reinforced by alterations to the natural environment through flood protection, drainage and irrigation interventions. High population pressure with increased urbanization and rapid industrialization is inducing land use change by taking up relatively flood-free agricultural land around the major cities. Nevertheless agriculture still dominates land management.

The land classification that has been established by MPO is used for the planning and assessment of flood management schemes and programmes. The classification is based on the experience of farmers and local populations, and collected through extensive fieldwork. Depending on flood depth in an average year, the framework classifies land in Bangladesh into five categories. Details of the MPO classification of land types are presented in Table 24.1.

- F0, or high land, is usually above normal flood level. Shallow flooding of less than 30 cm may occur occasionally in the rainy season. Most of the urban areas are considered as F0 land or high land.
- F1, or medium-high land, is land that is normally flooded to a depth of about 30–90 cm in the rainy season. Land that is shallowly flooded for a few hours at high tide each day is included.
- F2, or medium-low land, is land that is normally flooded to a depth of 90–180 cm in the rainy season. This type of land is also inundated for a longer period, as the water takes more time to recede.
- F3, or low land, is land that is normally flooded to depths ranging from 180 cm up to about 360 cm in the rainy season. The land becomes dry for all or most of the dry season.

Table 24.1 Classification of land types based on depth of flooding (after MPO 1986)

Land type	Description	Flood depth (cm)	Nature of flooding
F0	High land	<30	Intermittent
F1	Medium-high land	30–90	Seasonal
F2	Medium-low	90–180	Seasonal
F3	Low land	180–360	Seasonal
F4	Low to very low	>360	Seasonal/ perennial

- F4, or low to very low land, is land in depressions that normally stays wet for all or most of the year even during the dry season. Most of this type of land is deeply flooded to more than 360 cm in the rainy season. The important difference between very low land and other types of land is that it stays wet for all or most of the dry season, so cannot be used for most crops.

Figure 24.2 shows the broad distribution of these land types in the different regions in Bangladesh. It must be remembered, however, that a range of flood depths can occur even within the same village. In general, normal seasonal flooding is shallow in the northwest, west, east and south of the country, and is deep in the centre and north.

In modern flood protection and drainage projects embankments raised for flood control have been found to have caused appreciable changes in land types. Assessments of various FCDI project areas show that former F2 (medium-low land) and F3 (low land) were converted into F0 (high land) and F1 (medium-high land) (Brammar 2002). Since F0 and F1 provide areas where the richest diversity of crops can be grown, the production strategy concentrates on the requirement for these areas. Table 24.2 gives the percentage of land types prevalent in 1990.

Land Use Planning According to Land Types

Flood protection for major cities, important commercial and industrial areas and key transport and communication infrastructure is given the highest priority. Most urban areas in the country, which are protected by embankments, flood control and drainage works, are situated in F0 lands, or high lands. In fact in urban areas and especially for Dhaka the major economic benefits are measured as the increase in land values and protection of existing economic infrastructure. This is reflected in the design of the embankments surrounding the four major cities, Dhaka, Chittagong, Khulna and Rajshahi urban areas. A 100-year flood frequency is adopted as the design water level for urban settlements, whereas the embankments protecting agricultural areas are normally designed for a flood of 25-year frequency.

In rural areas the scenario is somewhat different. While even slight flooding can disrupt urban life and normal activities, inundations are not always unwelcome in rural areas. The Bengali language distinguishes between the normal floods of the rainy season, which are locally known as *barsha*, and the more harmful floods of abnormal depth and timing, which are termed *bonna*. The *borsha*, which occurs more frequently than *bonna,* is not considered (in rural areas) to be a hazard at all, but rather a necessity for survival. In fact for rice farmers too little water is a greater threat than too much. The relation between the cropping calendar and the seasonal flooding is documented in Figure 24.3. In different seasons of the year different varieties of rice dominate, adapted to the hydrological conditions of the respective season. The cropping calendar is not only adapted to the different seasons but also to the different levels of the land.

In rural Bangladesh, the farmers decide which crops to plant. In their decision-making, the relative profitability of crops and amount that can be safely harvested play a major and in many cases decisive role, though their decision-making is often constrained by resources. The total harvest depends on the agroecological environment, i.e. the land type according to flood depth. Depending on the type of land and soil characteristics the prevalent cropping calendars are described in Figure 24.3.

Although around 50 different crops are grown on the agricultural lands the calendar mainly evolves around rice plantation. This is because

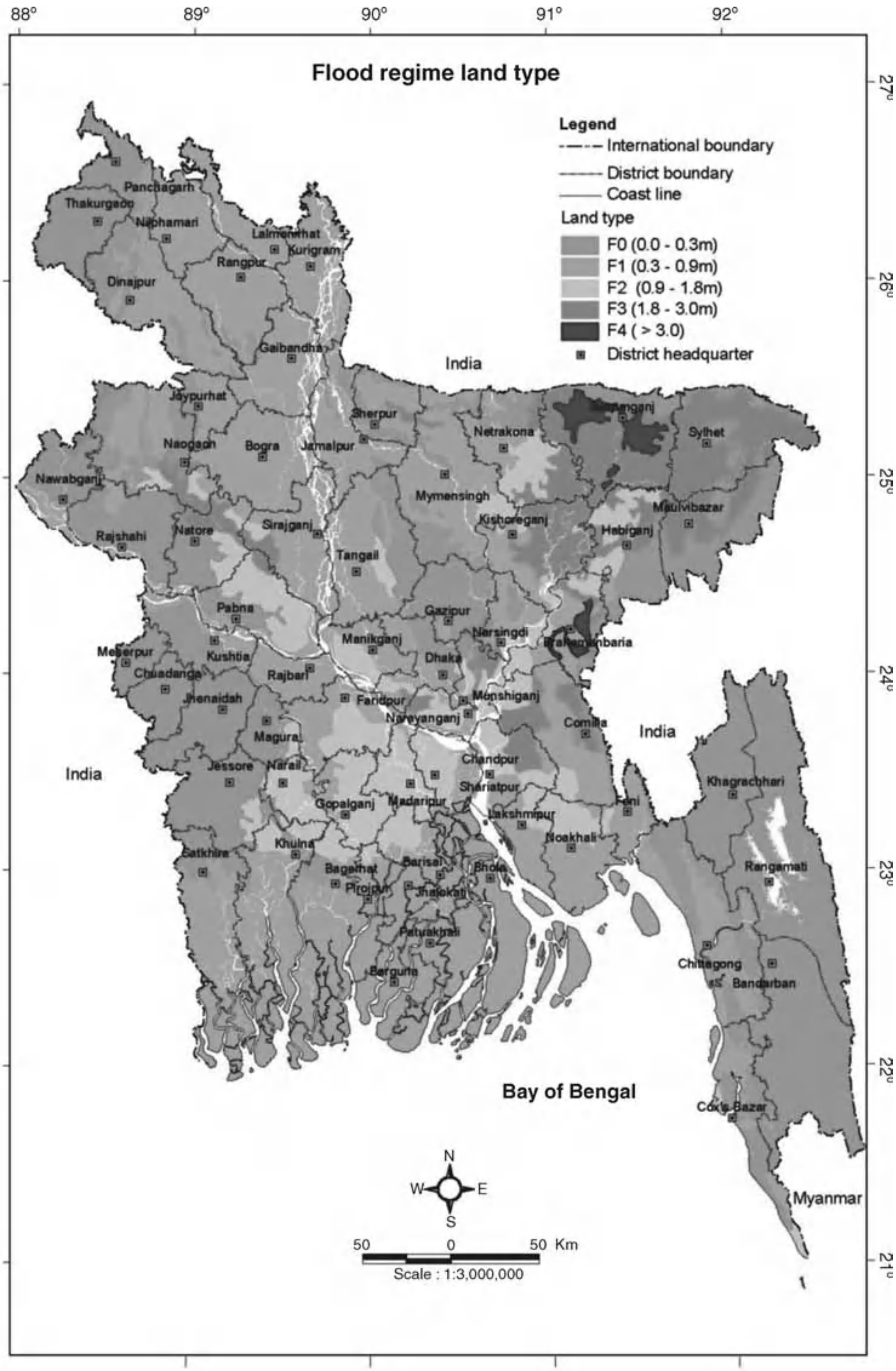

Fig. 24.2 Land type according to flood depth in Bangladesh. (See the colour version of this figure in Colour Plate section.)

Table 24.2 Land area (in sq. km) according to land types (from Alam *et al.* 1999)

Land type	Generalized area (1990)
F0 land	44,409
F1 land	32,708
F2 land	15,829
F3 and F4 land	14,311
Urban area	1539

the high rainfall and flooding conditions are particularly suitable for rice cultivation, and since rice is also the staple food, rice cultivation occupies around 80% of the cropped area in the country (Brammar 2002). There are generally two main cropping seasons (*Kharif* and *Rabi*) and three rice-growing seasons (*aus, aman* and *boro)*. The *Kharif* season further comprises an early part and later part.

• The *Kharif-I* season corresponds to the pre-monsoon and early monsoon (wet season). The principal crops are *aus*, paddy and jute. Aus plantation requires irrigation at the beginning of the season for land preparation and transplantation if the monsoon arrives late. After the monsoon arrives natural rainfall supplies the crop. This practice is observed to maintain the planting time. *Aus* is normally grown on the higher F0, F1 and F2 land types.

• The *Kharif-II* season refers to late monsoon and early post-monsoon. Transplanted *aman* paddy is the main crop in the Kharif-II season. *Aman* is sown during the monsoon and harvested post-monsoon. At a later stage of crop growth supplementary irrigation is often required to prevent yield losses in case of water shortage. Most varieties of *aman* are normally grown in higher F0, F1 and F2 land types while the deep-water varieties can be grown in the lower F3 lands.

• *Rabi* (dry season) consists of the cool winter months when crops such as wheat, pulses and oilseeds are grown, or the hot pre-monsoon when

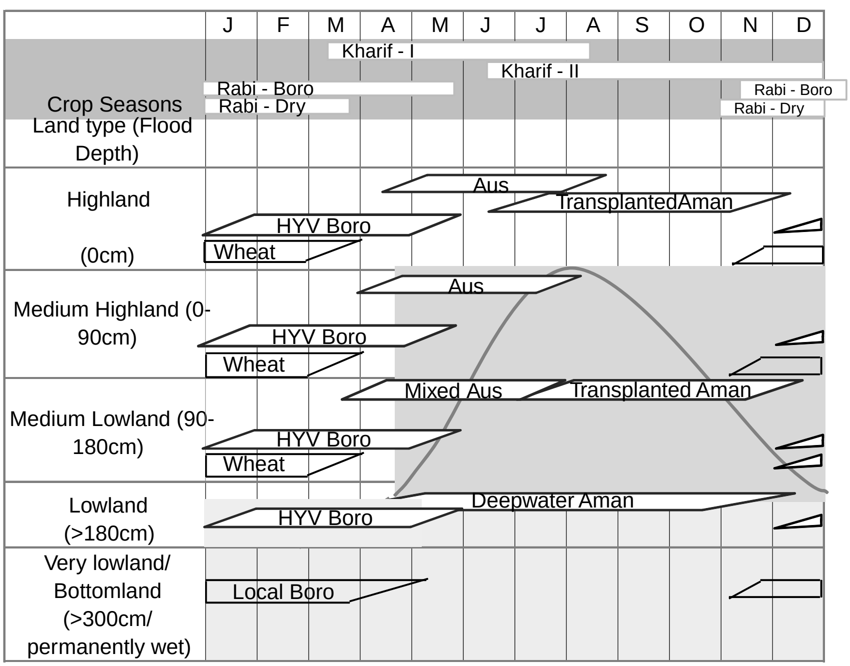

Fig. 24.3 Cropping patterns in relation to seasonal flooding, climate and natural disasters. From Brammar (2002).

summer fruits and vegetables are grown, or dry season rice varieties called boro. All *Rabi* crops need irrigation and can be grown on a variety of land types. Even in the permanently wet F4 lands, *boro* is grown locally.

The different cropping seasons usually overlap. As a result, farmers usually go for double cropping, i.e. two crops in one calendar year. Figure 24.3 shows a generalized cropping calendar for the entire country. But the schedule of plantation and harvesting differs slightly within the different hydroecological regions. The monsoon starts earliest in the northeast and moves towards the west, starting latest in the western part of the country. Similarly, the flooding pattern varies as the flood peak arrives earliest in the northern areas, and recedes latest in the southern coastal areas.

It is important to understand that the timing and duration of floods plays a major role in the cropping pattern. Early floods (in April/May) generally cause severe damage to *boro* crops. Severe damage to mature *boro* rice from flash floods is reported in the eastern foothill regions virtually every year. Moreover, excessive floods in June/July can harm standing *aus* crops and delay transplantation of *aman*. Impeded drainage of floodwaters causing water to stay in agricultural lands for more than five consecutive days can seriously reduce yield. Therefore, F0 (high land) and F1 (medium-high land) provide a wider range of options for intensification of cropping than the more deeply flooded land types. In very deeply flooded land types (F3 and F4) productivity is very low and cropping choices are also limited. However, pulses, oilseeds and other dry-land *Rabi* crops can be grown provided floodwaters recede before December.

Although agriculture receives the highest priority in water and land management in Bangladesh, fisheries are also given some consideration especially in *beel* and *haor* (perennial wetland) areas. Inland water bodies can be differentiated into open water bodies such as rivers, canals, *beels, haors* and floodplains, which produce capture fisheries, and closed water bodies such as ponds and lakes, which are increasingly being used for culture fisheries. The perennial water bodies are called *jalmahals* or fishing grounds. The annual flooding and post-flooding standing of water in the floodplains plays a vital role in the sustenance of fish stocks and the maintenance of species diversity in the open water fishery of the *jalmahals*. The floodplains comprise a rich ecosystem producing biomass that supports the major biological activities of fish. During the monsoon season, fish movement increases throughout the floodplain and from rivers into distributaries. Fishing seasons in the *haor* areas include the monsoon and post-monsoon months from mid-June to November, and also the dry and pre-monsoon months from December to early June. Fishing catch is highest during the monsoon when the migration and movements of fish take place. During November and December, dams are placed in the drainage canals to stop recession of floodwaters. Conversely, in February water levels in the *beels* are reduced through drainage to maximize the catch (Paul 1997).

Flood Management Practices in Bangladesh: Impacts on Land Use

In present-day Bangladesh, the FCD or FCDI projects dominate land and water management practices. The implementation agency for these projects is the Bangladesh Water Development Board. From an agricultural perspective the FCD/FCDI schemes are designed to:

- protect standing crops (*aus*, planted in March/April) against river floods;
- expand the area under monsoon rice (*aman*, planted in August/September) by excluding flood waters from the scheme;
- retain water in the schemes during the post-monsoon period.

Typical projects have three major components:

- embankments to control overbank spills;
- *khal* (or canal) closures to control entry of river floodwater;
- *khal* (or canal) regulators to control entry and drainage of floodwater.

Embankment construction has been the major activity in these FCD projects as part of the comprehensive flood and land management strategy. Embankments of a total length of more than 8300 km have been constructed since 1959 in the country. According to the Bangladesh Water Development Board (Ali 2002), based on types of infrastructure, location, topography and main flood management issues, the schemes can be classified into the following categories:

- *haor* (floodplain depressions in northeast part of the country) schemes;
- southeast coastal polders;
- southwest coastal polders;
- *beel* (floodplain depression) schemes;
- floodplain schemes.

Haor schemes

Haors are large saucer-shaped floodplain depressions located mostly in the northeastern region and covering about 25% of the entire region. There are altogether 411 haors comprising an area of about 8000 km^2 and dispersed in the districts of Sunamgonj, Sylhet, Moulvibazar, Hobigonj, Netrokona and Kishoreganj (Paul 1997).

The extreme flashy character of the rivers and extremely high rainfall causes frequent flash floods during the pre-monsoon period of April/May. These types of floods cause damage to standing *boro* and infrastructure. However, monsoon flooding and retention of water is imperative to sustain the complex hydroecological characteristics of the *haor* wetlands.

To protect the crops in the *haor* areas, the erstwhile *zamindar*s, with the participation of the local people, used to construct small dykes for early flood protection and irrigation. Based on these initiatives, since 1966, the BWDB has focused on the construction of about 2000 km of submersible embankments and additional structures in the *haor* region (BWDB 2008). Submersible river embankments provide protection from flash floods in the pre-monsoon, and during monsoon overtopping is allowed. The depressions can be inundated by 1.5–6 m of floodwater from May to November/December. After recession of floodwaters only the deepest parts remain wet.

Southeast coastal polders

In order to protect the coastal areas from regular tidal inundation and salinity intrusion, a total of 48 polders have been built since the 1960s under the Coastal Embankment Project (CEP). The southeast coastal polders are located in Chittagong and Cox's Bazaar, at the foot of the southeastern hilly regions. These polders are characterized by sea-facing embankments on one side and minor embankments for protection from river flooding from one or two sides. The schemes usually have relatively small parallel canals perpendicular to the sea with many cross-dams across the width of the canals. Numerous minor tidal sluices are located along the embankment for drainage. The drainage canals and tidal sluices have to drain out runoff from adjacent highlands in addition to retained rainwater within the polder area. Salt and shrimp production along with rice cultivation are the main economic activities within the polders (after Ali 2002).

Southwest coastal polders

The southwest region of Bangladesh is characterized by a flat, low-lying alluvial landscape interspersed by an extensive system of tidal rivers and streams and water-filled depressions locally called *beels*. The rivers are distributaries of the Ganges and the *beels* are actually oxbow lakes. The river system is highly active, carrying large concentrations of sediment, and the river waters are carried into the depressions with unobstructed high tides causing saline water intrusion.

Before the 1960s the locals built temporary embankments called *oshtomaisha gher*, meaning 'embankment of eight months'. Water was allowed to enter into the depressions during the monsoon when salinity in the rivers was low. The silt carried with the water dispersed in the depressions and was deposited during ebb tide. During the 1960s BWDB constructed a series of polders or closed embankments with numerous tidal regulators to reclaim elevation lands and check saline water intrusion. After more than a decade of increased productivity in agriculture, drainage congestion began to create serious waterlogging problems. The presence of polder embankments restricted tidal flow into the *beel* area preventing sedimentation in the low-lying areas. Moreover, reduced flows in the Ganges especially during the dry season reduced the flushing capacities of the rivers and canals. Siltation of drainage channels began to occur and by the 1980s many drainage canals became inoperative due to siltation rendering vast tracts of lands waterlogged all year round.

In the late 1990s, tidal basin management was identified and studied as an alternative solution to major regulators, proposed for relieving drainage congestion. Supported by public opinion, silt-laden waters are being allowed into designated tidal basins, thereby elevating the land by deposition of silt during ebb tide. The main features of Tidal River Management (TRM) are:

- tidal flow is allowed in the basin;
- tidal basin increases tidal volume;
- tidal basin stores flood water during flood event;
- sedimentation takes place during the long storage period acting as sedimentation trap;
- erosion and maintenance of rivers takes place.

Beel (floodplain depression) schemes

Several clusters of natural depressions, or *beels,* are situated in the southern part of the northwest region of the country. There are about 50 *beels,* having a surface area ranging from 25 to 1500 acres and depth varying from 0.3 to 3 m. These *beels* act as temporary flood retention reservoirs and as linkage channels between the parallel rivers, which are tributaries of the Jamuna. Since the 1970s a large number of polders consisting of one main river embankment and minor embankments have been constructed around these *beels* to protect rice from monsoon flooding. These schemes usually have one main drainage canal with one outfall. Higher land situated on two or three sides of the scheme results in runoff into the scheme (after Ali 2002).

Floodplain schemes

These schemes are characterized by high-level flood embankments on one or both sides of a river. The main source of flooding in the floodplain areas is bank overflow from the major rivers – the Ganges, Brahmaputra and Meghna – and their tributaries and distributaries during June to September. These floods are characterized by slow rise and fall extended over 10–20 days or more. Floodplains adjacent to the rivers are subjected to this type of flood every year, which affect about 30% of the country. Furthermore, high-intensity rainfall events of long duration cause local flooding. Average annual rainfall for the whole country is about 2400 mm. About 80% of annual rainfall occurs during June to September. In addition, during these months the main rivers and tributaries flow at high stage due to huge discharge from snowmelt and rainfall from catchments outside the country. High stage in the rivers impedes drainage, and the excess volume generated from rainfall causes local flooding.

Many canals run relatively parallel to the main rivers and are actually tributaries or distributaries of the rivers, providing drainage for the protected areas situated in the surrounding the country. Mostly, gravity drainage is available in these schemes, while in some cases pumped drainage is also provided particularly in areas of high economic value, such as Dhaka city.

Impacts and Issues

Infrastructure built for flood protection has, in general, provided enhanced safety and security to people, crops and livestock. The positive impacts include protection against early flooding (mainly submersible embankments), salinity exclusion, and reduction of monsoon flood depths. Protection has resulted in changes in land type, which consequently have brought about changes in land use patterns (Box 24.2). According to NWMPP (2000), the major agricultural impact of the hydrological and land type changes resulting from FCD schemes was on *Kharif* cropping practices within

Box 24.2 Case study: the Dhaka-Narayanganj-Demra project

The Dhaka Narayanganj Demra (DND) project demonstrates the impact of increased flood protection on land use and land management practices in Bangladesh. The project was originally designed as an irrigation project by the Bangladesh Water Development Board (BWDB) in the early 1960s to meet national objectives of achieving self-reliance in food grain production. The project was also conceptually identified to accelerate agricultural production by providing comprehensive Flood Control, Drainage and Irrigation (FCDI) facilities covering 56.79 km^2 of the Greater Dhaka District. The area is situated in close proximity to the capital Dhaka adjacent to a major national highway.

As a result, with the increase in flood-free land, over the last two decades the DND area has experienced a progressive change in land use from a potential agricultural area to an urban development area. Numerous industries of different categories have also sprung up within this project area. In 1990, a land use study by the Flood Action Plan (8A) found that 21.7 km^2 (38%) of the DND project area was urbanized with 31.7 km^2 (56%) land being used for agriculture (Japan International Co- operation Agency 1991). The present land-use pattern shows that 60% of the project area is urbanized while land used for agriculture has decreased to only 20%. Furthermore, the rapid urbanization and industrialization have caused an unchecked increase in population. Moreover, although the land is free from river floods, unplanned urbanization impedes drainage of rainwater, causing drainage congestion every year and forcing changes in land management practices.

the flood-protected areas. The dominance of paddy in cropping patterns increased; however, cropping intensities were rarely increased. Because of the more secure growing conditions in high (F0) and medium-high (F1) lands, a shift from B (broadcast) *aman* or *aus/aman* to TL (local transplanted) *aman* and transplanted HYV (high-yielding variety) *aman* was recorded. The change from F2 to shallow-flooded F1 has allowed the introduction of T *aman* in place of B *aman* (deepwater). Conversion of shallow-flooded F2 to very-shallow-flooded F0 allowed the introduction of HYV varieties (Brammer 2002). *Boro* expansion occurred where protection from early flash floods (mainly in the northeast) or irrigation was provided. Paddy yields improved as a result of the shift to higher yielding paddy types. As a result, data analysis shows that FCD schemes have been successful in raising agricultural production by 35% (Datta 1999).

In contrast, drainage congestion has been the most serious technical problem, reducing much of the efficiency of the FCD schemes and in many cases lowering agricultural productivity. According to NWMPP (2000), most projects concentrated on flood protection, with drainage receiving less attention. Drainage congestion has basically stemmed from inadequate planning and design of drainage canals and sluices. Appropriate consideration was not given to external impacts (e.g. higher flood levels outside polders, reduction of river flows downstream of cross-dams) and adverse internal impacts (such as siltation), and to the inclusion of mitigation measures (e.g. structural floodproofing to mitigate higher external flood levels). Inaccurate maps and other data deficiencies have also contributed to ineffective drainage.

Modern flood management practices have, on the whole, resulted in strongly negative impacts on the fisheries sector, with capture fisheries being the worst affected. Reductions in perennial *beels* and natural water bodies in deeply and regularly flooded areas due to late or controlled flooding have decreased fish habitat. In addition flood protection infrastructure has caused blockage of fish migration routes limiting reproduction. The production of culture fisheries has seen a slight improvement, but capture fisheries losses have far outweighed culture fishery gains. Moreover, such gains have not benefited the mostly poor capture fishermen. The inland open water fishery, which is a common property open-access natural resource available to the rural poor, and which provided more than 70% of the country's fish production only a couple of decades ago, is now in serious decline. In most cases capture fisheries losses were substantially greater than had been anticipated (NWMPP 2000).

The flood protection schemes carried out under the FCD projects brought about immediate results; however, later evaluations noted the rapid rate of decline in performance, especially in terms of operation and maintenance, of much of this infrastructure. In particular the increases in agricultural production (the main objective of the interventions) failed to materialize as predicted (Datta 1999). Inadequate operation and maintenance (O&M) has been a universal problem on virtually all public sector schemes evaluated, often hampering the sustainability of the schemes. Its causes include inadequate government funding, poor cost recovery, lack of beneficiary participation and technical difficulties.

Concluding Remarks

Land management in Bangladesh is determined mainly by the monsoon rainfall and the seasonal flooding, which affect the greater part of the country. Presently, these natural physical determinants are reinforced by alterations to the natural environment through flood protection, drainage and irrigation interventions.

Over the past three decades flood and water management in Bangladesh has gradually evolved to its current stage with huge investments being made in flood protection, drainage and irrigation schemes to reclaim and develop floodplain and coastal areas. Although these large projects still dominate management practices, considerable change and progress based on practical experience and 'lessons learnt', can be seen in the planning and implementation of these projects. Emphasis is on a more comprehensive approach with multi-objective and multi-sector planning, rather than single-objective (e.g. food grain self-sufficiency) planning. The generally better performance of small-scale interventions and private sector development has seen the reduced importance of large-scale government projects. Present integrated water and land use planning is set to focus on the development of floodplain zoning to accommodate necessary engineering measures and allocate space for habitation patterns, economic activities and environmental resources

Glossary

Aman	wet season rice crop
Aus	*rice varieties grown during pre-wet season*
Baor	*a floodplain depression of southwest Bangladesh*
Barsha	*normal flooding of the rainy season*
Beel	floodplain depression that may hold water perennially or dry out during the dry season
Bonna	infrequent, severe, hazardous flooding
Boro	dry season rice crop
Haor	saucer-shaped depression of floodplain located between two or more rivers jalmahal a perennial closed water body
Khal	canal
Kharif	either of two planting seasons: Kharif I, pre-monsoon and early monsoon; and Kharif II, late monsoon and early post-monsoon
Rabi	dry season
Zamindar	local landlords with responsibility for water resource management initiatives

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25 Goals, Institutions and Governance: the US Experience

GERALD E. GALLOWAY

> Floods have . . . devastated more families and communities in the United States than all other natural hazards combined
> **Interagency Floodplain Management Task Force**

In 1993, a major flood devastated the US Midwest causing nearly $20 billion in damages and the loss of over 138 lives. Over the next 15 years, the USA experienced several large regional floods, the catastrophe of Hurricane Katrina in 2005, and in 2008 a near repeat of the 1993 Midwest flood. These events demonstrated to the nation the shortfalls in its approach to reducing flood damages and the need to focus on the goals, governance mechanisms and institutions that shape that approach. It became obvious in the analysis of the current approach that the effort had become fragmented and there was a need to harmonize its many components. This chapter describes the US experience in evolving its current approach to flood management and steps that are being taken to deal with new challenges that have arisen.

Dealing with Floods: From Colonies to Katrina

The first formal efforts in the USA to deal with flooding occurred in the early part of the 18th century almost simultaneously along the banks of the Mississippi River at New Orleans and along the shorelines of colonial farms in southern New Jersey near the Delaware River estuary. While individual floodplain occupants had been dealing with local flooding by elevating their homes, moving to higher ground during peak flows, or using other adaptation techniques, the first formal projects were focused on keeping the flood away from property through use of earthen levees. These early efforts launched the new nation into a structural flood control paradigm. Throughout the 19th century and more than halfway through the 20th century, this focus on providing structural protection for those at risk continued. As the nation moved to the West and the growing population occupied more and more lands subject to flooding, the federal government was called on to take steps to mitigate the flood damages that were occurring. The Mississippi River Basin, draining 41% of the coterminous USA, was a central artery for commerce and a target for settlement and thus became the focal point of federal interest in dealing with floods (Fig. 25.1). In 1849, 1850 and 1860, federal legislation was enacted that gave title to swamplands across the country to the states in which they were located so that the states in turn could drain the wetlands, sell this new farmland to the public, and use the money to provide the same lands with protection against floods (USGS 2008). Thirty years later, faced with growing flood damages along the banks of the Mississippi River, the Congress established a Mississippi River Commission 'to take into consideration and mature such plan or plans and estimates' that will provide for navigation and prevent floods (Mississippi

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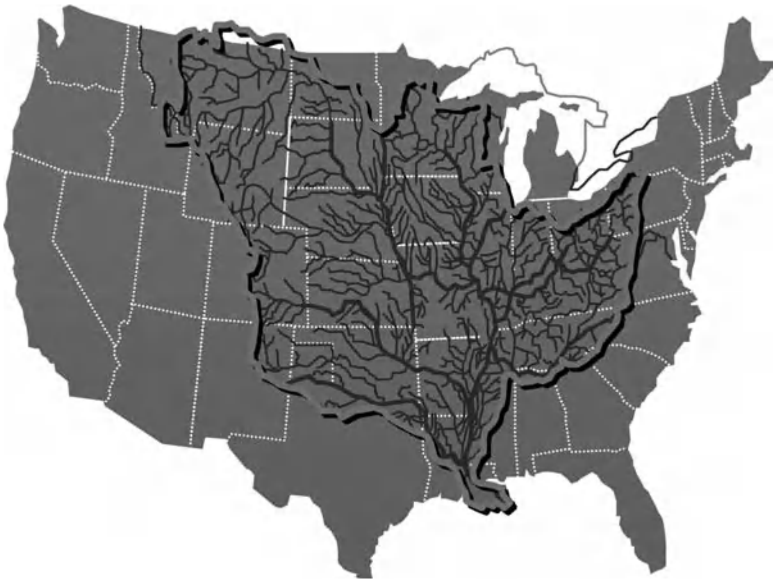

Fig. 25.1 The Mississippi River Basin. (See the calour version of this figure in Colour Plate section.).

River Commission Act 1879). The creation of the Commission followed a detailed study of possible flood control approaches conducted by the US Army Corps of Engineers (Corps) in the 1860s, which recommended the use of levees to channel the Mississippi and reduce flood damages (Humphreys and Abbott 1861). Because of arguments in the US Congress over the constitutionality of federal support of flood control, the Commission was not appropriated significant funds to deal with flood control. After disastrous floods in California and the Mississippi River Valley shortly after the turn of the century, the Congress authorized the Corps to increase its support of flood control activities in both the lower Mississippi Valley and in the Sacramento River in California, and increased the appropriations for these efforts (Arnold 1988).

In 1927, a major flood wreaked havoc on the lower Mississippi River Valley and brought national attention to the risk being faced by those who lived along the river, farmed the rich alluvial lands, and populated the major cities. Hundreds of thousands were driven from their homes for months and the economic losses were staggering for those in the Southern states. Congress reacted by passing the Flood Control Act of 1928, which assigned the Corps and the Mississippi River Commission responsibility for development and operation of a flood control system for the lower Mississippi Valley. Eight years later, following major flooding across the nation, the Congress passed the Flood Control Act of 1936, which indicated that 'the Federal Government should improve or participate in the improvement of navigable waters or their tributaries including watersheds thereof, for flood-control purposes if the benefits to whomsoever they may accrue are in excess of the estimated costs, and if the lives and social security of people are otherwise adversely affected.' The Act gave the Corps responsibility for controlling floods around the nation using techniques that it had put into use over the previous decades – levees, floodways and dams.

In the 1960s, spurred on by the work of the geographer Gilbert F. White and with the continuing rise in flood damages, many began to advocate use of non-structural approaches to reducing flood damages. In 1968, the Congress authorized the establishment of the National Flood Insurance Program (NFIP), which would offer flood insurance for purchase by floodplain residents (National Flood Insurance Act 1968). In 1970, the President signed the National Environmental Policy Act (NEPA), establishing a clear US position on protection of natural resources including those in the riverine area. The birth of the NFIP and the passage of NEPA increased national interest both in using non-structural flood damage reduction tools (as opposed to controlling floods) and in protecting riverine ecosystems that had been badly damaged by settlement and flood protection activities.

The 1993 upper Mississippi Basin Flood brought renewed attention to the floodplain. A 1994 White House report emphasized the need for new approaches to reducing the risk to the people and property in the floodplain (Interagency Floodplain Management Review Committee 1994). In 1996, the Corps issued instructions to its field activities directing that future projects be developed using risk concepts (US Army Corps of Engineers 1996). Following Hurricane Katrina in 2005, the Corps formally shifted its focus from flood damage reduction to flood risk management.

Over a period of three centuries, the US approach to dealing with flood damages gradually shifted from total reliance on structural 'flood control' activities, through a mix of structural and non-structural approaches, to, most recently, a movement towards broader-based flood risk management.

Setting a Direction

The direction taken in a flood management program is driven by its explicit and implicit goals. In the USA, these goals appear primarily in legislation and are influenced by the division of responsibilities defined in the US Constitution between the federal government and the states of the Union.

Floods are natural events and not only replenish alluvial soils, substantially increase the yield of the land, and sustain rich habitat for natural systems, but also inflict substantial damages on human activities in the floodplain. Nations historically have been forced to trade off economic and social development with potential damages that result from flood management activities. While nations are developing, the emphasis seems to be on the economic and social. At some stage, there comes recognition of the environmental consequences of the activity and the increase in damages caused by populating high-risk areas in the floodplain. In the early stages of development in the USA the emphasis was on flood damage reduction, mitigating the impacts of periodic flooding at the local level through a variety of methods. Over time, as the central government became more of a factor, the flood management goals became more codified.

The Tenth Amendment of the Constitution of the USA indicates that 'The powers not delegated to the United States by the Constitution, nor prohibited by it to the States, are reserved to the States respectively, or to the people' (US Constitution 1791). During the first century of the nation's existence, this amendment played heavily in defining responsibilities for flood management activities, as most federal legislators did not see a federal responsibility to do more than understand the science behind flooding and possibly the threats that flooding posed. In establishing the Mississippi River Commission, the Congress also established federal goals to 'improve and give safety and ease to the navigation [of the Mississippi River]; prevent destructive floods; promote and facilitate commerce, trade, and the postal service...'. As previously indicated, during its initial years, the Commission was funded to deal more with navigation and flood control, as the Congress continued to worry over its authority over flood control. Since providing safety and ease of navigation required the Commission to maintain a defined channel through use of dikes, levees, bank protection and dredging, it was frequently difficult to determine where navigations stopped and flood control began. Gradually, but without explicit direction from the Congress, the Commission became more and more involved in flood control activities. The 1917 Flood Control Act for work on the Mississippi and Sacramento Rivers helped to further define a growing federal interest and permitted proponents of flood control to argue that it did not represent subsidies to local interest for development, but was necessary to alleviate the hardships of flooding (O'Neill 2006). The 1928

Flood Control Act moved federal interest to a higher level and firmly established a goal of preventing recurrences of the 1927 event. Up until 1928, local and state governments carried the primary responsibility for carrying out flood control. The 1928 Act found that '...in view of the extent of national concern in the control of these floods in the interests of national prosperity, the flow of interstate commerce, and the movement of the United States mails; and, in view of the gigantic scale of the project, involving flood waters of a volume and flowing from a drainage area largely outside the States most affected, and far exceeding those of any other river in the United States, no local contribution to the project herein adopted is required.' With this Act, the Congress established a federal role in flood control and related the objectives of flood control to the economic well-being of the nation as a whole.

The 1936 Act further strengthened the federal role finding that '...it is the sense of Congress that flood control on navigational waters or their tributaries is a proper activity of the Federal Government in cooperation with States, their political sub-divisions and localities...that improvements of rivers and other waterways, including watersheds thereof, for flood-control purposes are in the interest of the general welfare; that the Federal Government should improve or participate in the improvement of navigable waters or their tributaries including watersheds thereof, for flood-control purposes if the benefits to whomsoever they may accrue are in excess of the estimated costs, and if the lives and social security of people are otherwise adversely affected.' While further defining the federal goal of improving the general welfare of the nation, the Act did restore responsibilities of the local and state governments for providing the lands easements and rights-of-way necessary to carry out the flood control activities. From 1936 through 1968, controlling floods clearly was the goal of federal, state and local governments.

Economic and social concerns after World War II focused attention on the growing cost of flood control activities and the failure of the work to date to significantly diminish annual losses. Gilbert White's thesis that losses could be avoided by sensible occupation of the floodplain resonated with many individuals, and one agency, the Tennessee Valley Authority, began to apply floodplain management techniques to control unwise development. Following major property damage during an East Coast hurricane and submission to Congress of the report by a team led by Gilbert White, the Congress once again turned its attention to the floodplain noting that '...despite the installation of preventive and protective works and the adoption of other public programs designed to reduce losses caused by flood damage, these methods have not been sufficient to protect adequately against growing exposure to future flood losses.' In passing the National Flood Insurance Act the Congress established '...as a matter of national policy, a reasonable method of sharing the risk of flood losses...through a program of flood insurance which can complement and encourage preventive and protective measures.' Thus, in 1968, a new goal began to be defined – sharing the risk of flood losses. Use of 'preventative and protective measures' joined flood control as a method of achieving national flood management goals. The 1968 Act and a supplementary Act in 1973 made flood insurance available to communities that wished to participate in a national program, if in turn the communities would establish controls over future development in the floodplain. The 1973 Act also required that anyone living in the 100-year floodplain and obtaining a structure mortgage that was federally insured (most mortgages) would have to purchase flood insurance on the property.

The 1960s were a time for attention to environmental issues. The United States and the world became more cognizant of the environmental degradation that had taken place over the previous decades and the inability of the environment to further sustain such intrusions. In 1970, NEPA also established goals, declaring that 'The Congress, recognizing the profound impact of man's activity on the interrelations of all components of the natural environment, ...declares that it is the continuing policy of the Federal Government, in cooperation with State and local governments, and other concerned public and private organizations, to use all practicable means and measures, including financial and technical assistance, in a manner calculated to foster and promote the general welfare, to create and maintain conditions under which man and nature can exist in productive harmony, and fulfill the social, economic, and other requirements of present and future generations of Americans.' The environmental movement also led to Congressional enactment of other legislation designed to turn the philosophy into actions. Laws were passed dealing with protection of endangered species, attainment of clean water goals, preservation of historical resources, and preservation of coastal areas. Actions to carry out flood control activities quickly became embroiled in conflicts with the newly enacted environmental legislation.

In 1977, shortly after taking office, President Jimmy Carter issued an executive order establishing as a goal the avoidance of the long- and short-term adverse impacts of floodplain occupance and modification. To avoid federal support of floodplain development, he directed that each federal agency '...shall provide leadership and shall take action to reduce the risk of flood loss, to minimize the impact of floods on human safety, health and welfare, and to restore and preserve the natural and beneficial values served by floodplains in carrying out its responsibilities' (Carter 1977). This action defined the President's goal of accomplishing activity in the floodplain in a manner that would not further perpetuate flood losses or harm the environment.

During the subsequent Administration, President Ronald Reagan was responsible for actions that affected national goals and impacted the relationship between federal and state governments with respect to flood control. In 1983, he established that 'The Federal objective of water and related land resources project planning is to contribute to national economic development consistent with protecting the Nation's environment' (US Water Resources Council 1983). In 1986, at his behest, the Congress passed The Water Resources Development Act (1986) requiring that states and local communities seeking federal flood control (and other water development) projects would be required to share in the cost of these projects. These two actions had a distinct impact on how flood damage reduction projects were developed over the next decades. Prior to the 1983 action, national economic development and environmental quality were co-equal objectives, and regional economic development and other social effects were also to be considered. Limiting the justification to economics disadvantaged flood damage reduction projects where the principal benefit rested in protection of lives or where the protected properties were of low value, typically the case with an economically disadvantaged population. Cost sharing led to least-cost alternatives to ensure that the solutions proposed met the fiscal capabilities as opposed to the flood damage reduction needs of the communities. The combination of these two activities implicitly shifted the goal of flood management from the general reduction in flood losses to dealing with only those that were affordable to the local communities and produced large economic benefits. It moved the de facto flood standard for levees towards 100-year protection as opposed to a much higher standard project flood level, which had been the choice of the Corps under full federal funding.

Neither the US Congress nor the Corps has defined an explicit goal for management of the nation's floodplains. The 1994 White House study of the 1993 Mississippi River flood pointed out the lack of clear flood damage reduction policy. National Research Council studies have noted the lack of a comprehensive national water policy (National Research Council 1999, 2002). Dialogues held by the American Water Resources Association at the request of federal agencies in 2002, 2005 and 2007 also have reported to the President the absence of federal or national water policies (American Water Resources Association 2007).

Acceptance by the Congress or the Administration of the concepts of Integrated Water Resource Management (IWRM) would represent a policy statement but no move has been made in that direction by either body. The Congress recently held hearings on watershed planning but took no action on development of any policies in this regard.

It is clear that leaders at various levels of government are concerned by growing flood damages and their impact on not only the economy but also the fabric of society. The experience of Hurricane Katrina and more recently the Midwest Floods of 2008 illustrated the pervasive nature of floods that break apart families, devastate communities and stall economies. While there is this concern, there is no agreement among these various levels of government as to what the goal of activity in the floodplain actually should be, and as a result there is program drift and a lack of support for developing comprehensive approaches and long-term solutions. Without clearly defined policies that set clearly vetted and consensus-built goals for the nation, it will be difficult for the USA to develop a cohesive and effective long-term strategy to deal with floods.

The development by the Association of State Floodplain Managers Foundation of a vision for the floodplain of 2050 provides an excellent example of what is needed to support policy development and the definition of clear goals 25.1.

Institutions

Flood management activities in the USA are the province of many institutions and several levels of government, and it is this variety of organizations that makes flood management so challenging. The most important institutions are the national, state and local governments. Between the state and the individual citizen there may be several

Box 25.1 *Floodplain Management 2050* (Association of State Floodplain Managers Foundation 2008)

In November 2007, the Association of State Floodplain Managers Foundation conducted a two-day forum to discuss with US and UK flood management experts their vision of what would be a desirable state for the floodplains of 2050. The consensus view of the attendees is reflected below:

Imagine the United States in 2050… *in spite of a growing population and a changing climate, both flood risk and land and water resources are being managed towards sustainable outcomes.*

- *The nation views land and water as precious resources, and therefore protects the natural and beneficial functions of floodplains, wetlands, and coastal areas.*
- *Because naturally flood-prone areas have been preserved – and restored where necessary – a maximum amount of natural mitigation of flooding takes place continually. A wide network of green infrastructure protects natural resources and functions and provides open space and recreational opportunities.*
- *Integrated water management is an accepted practice.*
- *All new development is designed and built so that it has no adverse impact on flood levels, sedimentation, erosion, riparian or coastal habitat, or other community-designated values.*
- *The free market strongly favors sustainable development, so flood-prone construction rarely occurs.*
- *Private and public losses due to floods are indemnified through a government-backed but private system of universal insurance coverage that encourages mitigation of damage.*
- *Management of floodplains is funded through fees charged for development impacts, a highway trust fund, or other secure sources.*
- *Risk communication has become advanced enough that local decision-making is well informed. Individuals and households understand both the risks and resources of natural flooding processes.*
- *Policy decisions about the use of land and water resources are based on sound data, science and models.*

layers of government, with the structure of the sub-state organizations differing across all 50 states. The structures of government at the federal and state level are generally the same, with each having an executive branch headed by an elected official (the President at the federal level and Governors at the state), a legislature with two houses (Senators and Representatives), and a judiciary – various courts to deal with civil and criminal issues and to address challenges to the legality/constitutionality of actions taken by the government and others. This structure provides for checks and balances among the different governmental entities at each level, and while useful in ensuring that no one arm of government becomes dictatorial, it creates tensions among these elements and makes rapid change more difficult.

At the state and federal level, and many times in the governance of large cities, the actual operations of the executive branches are carried out by agencies whose responsibilities are tied to a specific function, such as agriculture, commerce, etc. (Fig. 25.2). Within the Congress and state legislative bodies, committees, generally paralleling the responsibilities of the executive agencies, carry out the detailed work. In neither the executive nor the legislative branches are responsibilities for water or flood management found in a single organization, but rather are scattered throughout many agencies and committees.

Government at the local level varies considerably by state. Most have intermediate level organizations, counties or townships, between the states and municipalities, with the specific responsibilities of the municipalities and intermediate organizations defined by the constitutions of the states.

Water-related organizations are many and operate under charters developed by the federal government, states and local governments. They range from river basin organizations with formal authorities to water and sanitation districts, flood control districts, public and private utilities, and levee and drainage boards, which typically operate local flood works. In some cases, at the city and village level, flood works are carried out directly by these organizations.

Federal and state governments raise funds through taxes to carry out their functions. Lower levels of government may also raise funds through taxes when they are authorized by state governments. Special organizations, such as levee and drainage districts, are frequently given the authority to assess those receiving the benefits of their activities for the costs of carrying out their duties, although typically the strict controls placed on these assessments by the granting body does not necessarily guarantee that they will be authorized to raise sufficient funds to deal with the challenges they face. These special organizations can also receive funding directly from the federal or state governments. In the case of flood management, many times, local governments become sponsors for flood management projects in their jurisdiction and participate in the funding of these activities.

With so many independently operating entities, the challenge becomes coordination. The report of the White House committee following the 1993 Mississippi Flood found that:

> The division of responsibilities for floodplain management activities among and between federal, state, tribal, and local governments needs to be clearly defined. Within the federal system, water resources activities in general and floodplain management in particular need better coordination. State and local governments must have a fiscal stake in floodplain management; without this stake, few incentives exist for them to be fully involved in floodplain management. State governments must assist local governments in dealing with federal programs. The federal government must set the example in floodplain management activities
>
> **Interagency Floodplain Management Review Committee (1994).**

With the federal government playing a dominant role in floodplain management through the development of structural approaches and the management of the National Flood Insurance Program, the states and local governments, as well as the special bodies dealing with floodplain management, appear to have assumed that the federal

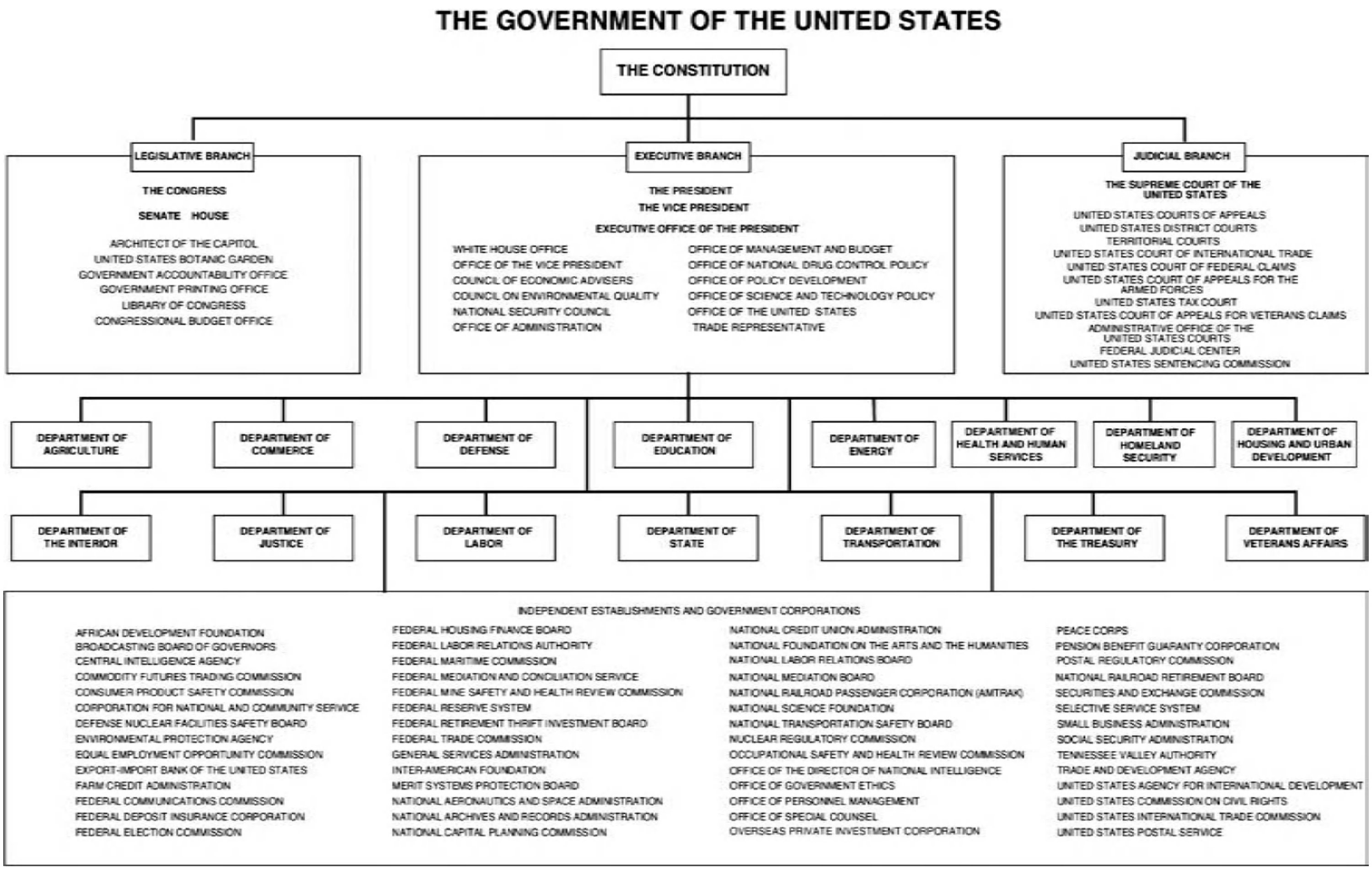

Fig. 25.2 The US (Federal) Government.

government was ultimately responsible and that the state and local roles were secondary. In most cases, state and local initiatives are very limited. The report recommended that responsibility needed to be shared at all levels to include assumption of some responsibility by the individuals who lived and worked in the floodplain. It recommended legislation that would define the responsibilities at each level of government. No action was taken on these recommendations.

Translating Policies into Action

Policy development and goal setting provide direction. Laws establishing policy are followed by other laws that authorize government activities, define implementation procedures in general terms, and provide for the fiscal support of the activities. These laws in turn are taken by agencies and translated into regulations, operational guidance and funding decisions that theoretically lead to actions that support the initial policies. The regulations and operational policies not only represent agency interpretation of the intent of the laws but also reflect guidance given to the agencies by the Administration and by the Congress outside of the legislation itself. In dealing with flood management, movement forward is associated with authorization and funding of projects, and programs and regulations that affect land use. Again, the same situation exists at state and frequently at local level.

Unfortunately, the governance activities are carried out by myriad organizations at the several levels of government and the coordination of these activities is often missing. Laws and guidance prepared by a congressional committee dealing with water quality might well impinge on floodplain management, but typically would not be coordinated with legislation being prepared by a committee dealing with flood control or dealing with flood insurance. Governing organizations at state and local level are often oblivious of laws and regulations and the implementing guidance at the federal level. As the number of these semi-independent actions grows, the probability for achieving effective flood management diminishes significantly.

The federal approach to dealing with flood management is defined by the 'Unified National Program for Floodplain Management', a document prepared in 1995 by an interagency task force at the federal level (Federal Interagency Floodplain Management Task Force 1994). The Unified National Program lays out a strategy for dealing with flood damage reduction but has no statutory authority to require agencies at the federal level or the states to follow this strategy. As a result, it is largely ignored and has been supplanted by implementation of individual pieces of legislation and guidance issued by multiple agencies across the federal government.

In the 1936 Flood Control Act, the Congress established the policy that flood control is a proper responsibility of the federal government. As a result, levees, flood walls, floodways and dams with flood control storage are typically funded at the federal level and require both authorization by the Congress and the subsequent support of funding for the projects by both the Congress and the Administration. When flood problems arise and are brought to its attention, the Congress directs that the problems be studied. On receipt of study reports from the Administration indicating that there are solutions to the problems and that their pursuit is feasible and desirable, the Congress then passes legislation authorizing the recommended projects for construction. In a separate process, the Congress subsequently might or might not provide funding to support the construction of the projects and the Administration might or might not place them on its priority list for action. Each project also requires a local sponsor, who would be responsible for providing the lands easements and rights-of-way for the project, and subsequent to 1986, a share in the cost of the projects. In some cases, the states were involved in these actions between local sponsors and the federal government; in others, they were essentially observers. Following completion of the construction of most flood protection structures such as levees and floodwalls, the local sponsor must assume responsibility for operation and maintenance of these facilities. However, states, because the federal government leads the program, have taken an almost hands-off approach to oversight of these structures. In 2006 only two states had inventories of levees within the state, and few provided any inspection or other oversight of the maintenance programs, leaving this to the federal government.

Smaller flood damage reduction structures are often funded by state or local agencies, or in some cases private groups, including businesses and developers. In most cases, coordination or approval of this activity by the federal or state government is not required (other than to obtain permits when the project might affect water quality or endangered species). While a permit is normally required to build a structure such as a home and office, no such construction permit is typically required for a levee that does not impact water quality or endangered species.

Guidance from the Administration and Congress also can affect what flood management activities take place and the manner in which they are carried out. In 1983, the Administration issued *Economic and Environmental Principles and Guidelines for Water and Related Land Resources Implementation Studies* (P&G), which established that the '. . .Federal objective of water and related land resources project planning is to contribute to national economic development consistent with protecting the Nation's environment' (US Water Resources Council 1983). This action, taken without coordination with the Congress, effectively removed life-safety and social effects as reasons for project development and placed the protection of the environment, which had been a coequal objective, in a secondary position. This led a 2000 National Research Council study to find that the P&G used for flood damage reduction studies '. . .emphasize direct economic damage reductions and the costs of alternatives' and recommended that:

> To ensure that the Corps's flood damage reduction projects provide adequate social and environmental benefits. . .the Corps [should] explicitly address potential loss of life, other social consequences, and environmental consequences in its risk analysis. Furthermore, the Corps's risk analysis should not be limited to structural alternatives such as levees, dikes, and dams. Nonstructural alternatives such as warning systems and zoning regulations should also be considered, both separately and in conjunction with structural alternatives.
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> **National Research Council (2000).**

In 2007, the Congress passed a Water Resources Development Act directing the Corps to prepare revisions to P&G that would include consideration of public safety as well as environmental sustainability. (While the P&G applied to several federal agencies, Congress only directed the Corps to prepare a new version for its use. It is not clear what role the Administration will play in evaluating any proposal by the Corps to the Congress.)

In 1968, the passage of the Flood Insurance Act indicated national support for this nonstructural approach to flood management. The intent of the act was to assist flood-prone citizens to obtain insurance (so as to reduce the fiscal outlays by the government) and to require local governments participating in the flood insurance program to control development in the floodplain with an overall goal of reducing national flood losses. By 2008, there were over 20,500 communities participating in the program and, as a result, controlling development in the 100-year floodplain. The Federal Emergency Management Agency (FEMA), which operates the NFIP, estimates that this latter component of the NFIP reduces flood losses annually by $1 billion (Maurstad 2008). As an incentive to the communities, FEMA developed a Community Rating System (CRS) that assesses the effectiveness of the local governments' efforts to meet FEMA objectives. Communities are given credit for, among many factors: requiring freeboard above the 100-year minimum structure elevation; taking actions to discourage development in the floodplain; and enhancing or protecting the natural and beneficial function of the floodplain. FEMA, in turn, reduces the insurance premium for those in participating communities as their CRS ratings rise.

It has been well recognized that wetlands are critical components of the riverine and coastal floodplains and provide substantial goods and services to the public at large through flood storage, aquifer recharge, storm buffering, attenuation of water pollution, and provision of habitat for ecosystems. The Federal Water Pollution Act Amendments (1972) legislated that those proposing actions that might disturb wetlands would be required to obtain a permit from the federal government. This legislation was translated by the Administration into guidance indicating that, should the proposed action 'have an unacceptable adverse effect on municipal water supplies, shellfish beds and fishery areas (including spawning and breeding areas), wildlife, or recreational areas' a permit could be refused. Under some conditions, minor wetland losses could be mitigated by construction of other wetlands or replacing affected wetlands with wetlands from 'wetland banks'.

While the federal government was developing its approach to activity in the floodplain, many communities in the floodplain determined that the way to growth (and increased tax revenue) was through development of floodplain lands. If the communities were in the NFIP and thus were required to limit development below the 100-year flood elevation, they either sought federal support for construction of protective levees or built levees using developer or local funds. In many cases, existing agricultural levees with low levels of protection were improved to provide protection against the 100-year flood. In other cases, communities that were not in the NFIP supported large-scale urban development in the floodplain and then sought federal support for construction of levees at the 100-year level. Communities were able to economically justify such levees because soon after the NFIP was initiated, FEMA, under pressure from developers, ruled that if an area was behind a levee that provided protection to the 100-year level, those living behind the levee would no longer be required to obtain flood insurance. Under the P&G, in developing a benefit-cost analysis to justify a project, the Corps could consider the elimination of the requirement for insurance as an economic benefit to the community. Such an approach obviously ignored the residual risks faced by those behind levees when the levees were overtopped or failed prior to overtopping.

Such actions by local communities occur because the incentive structure for communities is far different than that for the federal or state governments. Advantages accrue to the communities with increased development because, in most cases, they reap the benefits of a higher tax base with more development but do not bear the costs of public assistance and recovery when areas they have permitted are flooded. California recently passed legislation that requires cities and counties to contribute their 'fair and reasonable share' of the costs of property damage when they increased the state's liability by 'unreasonably approving new development in a previously undeveloped area' (Flood Liability Act 2007).

Few new federal, state or local flood management projects are developed within the context of either watershed planning or integrated water resources management (IWRM). During the decades immediately following the 1936 Act, most authorized projects were part of a larger number of projects that supported watershed approaches to solving the flood problems of the region. Over the last five decades, most projects have had little association with a watershed plan. Two notable exceptions are projects of the Tennessee Valley Authority, which continues to guide work along the Tennessee River and its tributaries, and the Mississippi River Commission, which was given responsibility by Congress for developing flood protection for the lower Mississippi River Valley in a comprehensive, integrated manner. While watershed planning is a clear goal of most federal and state water agencies, and the concept of IWRM has been generally accepted by water resources professionals, the stovepipe organizational structure at both the federal and state level tends to force agencies to operate independently of each other. This separation is exacerbated by structures in the Congress and state legislatures that maintain these same stovepipe structures. In addition, the Congress has long focused on authorization and approval of projects without any consideration of their watershed context or a funding of watershed studies that might provide a basis for projects.

Learning from the US Experience

By their nature, democratic nations create broad political involvement in government actions. This involvement makes development of a well-coordinated national (not federal) flood management program very difficult. As the USA has grown in population and territory, it has had to deal with the challenge of flooding and the damages that result from this flooding. Clearly the nation and its subordinate jurisdictions need to have goals and objectives that would lead the nation into an efficient and effective program to deal with this flooding.

At the federal level, and within most states, no one agency or committee is responsible for flood management. Prior to 1983, within the office of the President, there was a Water Resources Council, comprising the heads of the federal agencies with responsibilities for all aspects of water. The Council assumed this coordinating function and actually created the Federal interagency task force that prepared the Unified National Program. However, without a Council, the task force lost its gravitas, as did the Unified National Program, and as a result there has been no formal coordination of the Program in over 15 years. Shortly before Hurricane Katrina, the Corps and FEMA began an effort to rejuvenate coordinating mechanisms and, following Katrina, to move to a new paradigm in flood management by shifting from flood damage reduction to flood risk management. Although not formally endorsed by either the Administration or the Congress, the program is proceeding on an informal basis. In carrying this out both FEMA and the Corps have emphasized that neither insurance nor structures eliminate the total risk to those who live and work in the floodplain.

This new approach, spurred by the lessons learned in Katrina and the floods subsequent to Katrina, is bringing together the different levels of government and is focusing attention on those disconnects that exist among the various government mechanisms and the need for better coordination within the federal government and among the federal, state and local governments. The 2007 American Water Resources Association (AWRA) Water Policy Dialogue reported to the President and the Governors of all the states the need for this better coordination and noted that it was the sense of the dialogue participants that the center of gravity (focus of responsibility) for water issues was shifting from the federal to the state and local level (AWRA 2007).

It is important that flood professionals appreciate the difficult institutional and governance issues that are part of the environment in which they operate. It is highly probable that the interfaces among the many participants in flood management will continue to remain complex. Given that situation, it is critical that all recognize the importance of inter- and intra-agency coordination and cooperation and, while hoping for a nirvana that would bring greater harmonization of the multiple aspects of flood management, continue to recognize the challenges they face and, once seen, to deal with them.

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