COMPREHENSIVE Assessment of water management in agriculture

Research Report 17

Water Requirements of Floodplain Rivers and Fisheries: Existing Decision Support Tools and Pathways for Development

A. H. Arthington, E. Baran, C. A. Brown, P. Dugan, A. S. Halls, J. M. King, C. V. Minte-Vera, R. E. Tharme and R. L. Welcomme

Griffith UNIVERSITY
WorldFish CENTER
IWMI International Water Management Institute
SOUTHERN WATERS
MRAG
UNIVERSITY OF CAPE TOWN IYUNIVESITHI YASEKAPA • UNIVERSITEIT VAN KAAPSTAD
Imperial College London

The Comprehensive Assessment of Water Management in Agriculture takes stock of the costs, benefits and impacts of the past 50 years of water development for agriculture, the water management challenges communities are facing today, and solutions people have developed. The results of the Assessment will enable farming communities, governments and donors to make better-quality investment and management decisions to meet food and environmental security objectives in the near future and over the next 25 years.

The Research Report Series captures results of collaborative research conducted under the Assessment. It also includes reports contributed by individual scientists and organizations that significantly advance knowledge on key Assessment questions. Each report undergoes a rigorous peer-review process. The research presented in the series feeds into the Assessment's primary output—a "State of the World" report and set of options backed by hundreds of leading water and development professionals and water users.

If you are interested in submitting a report for inclusion in the series, please see the submission guidelines available on the Assessment website or send a written request to: Sepali Goonaratne, P.O. Box 2075, Colombo, Sri Lanka.

Comprehensive Assessment of Water Management in Agriculture Research Report 17

Water Requirements of Floodplain Rivers and Fisheries: Existing Decision Support Tools and Pathways for Development

A. H. Arthington, E. Baran, C. A. Brown, P. Dugan, A. S. Halls, J. M. King, C. V. Minte-Vera, R. E. Tharme and R. L. Welcomme

International Water Management Institute
P O Box 2075, Colombo, Sri Lanka

The Comprehensive Assessment (www.iwmi.cgiar.org/assessment) is organized through the CGIAR's Systemwide Initiative on Water Management (SWIM), which is convened by the International Water Management Institute. The Assessment is carried out with inputs from over 100 national and international development and research organizations—including CGIAR Centers and FAO. Financial support for the Assessment comes from a range of donors, including core support from the Governments of the Netherlands, Switzerland and the World Bank in support of Systemwide Programs. Project-specific support comes from the Governments of Austria, Japan, Sweden (through the Swedish Water House) and Taiwan; Challenge Program on Water and Food (CPWF); EU support to the ISIIMM Project; FAO; the OPEC Fund and the Rockefeller Foundation; Oxfam Novib and CGIAR Gender and Diversity Program. Cosponsors of the Assessment are the: Consultative Group on International Agricultural Research (CGIAR), Convention on Biological Diversity (CBD), Food and Agriculture Organization (FAO) and the Ramsar Convention.

The authors: A. H. Arthington, Australian Rivers Institute and eWater CRC, Faculty of Environmental Sciences, Griffith University, Nathan, Queensland 4111, Australia; E. Baran, WorldFish Center (formerly ICLARM), P. O. Box 500 GPO, 10670 Penang, Malaysia; C. A. Brown, Southern Waters Ecological Research and Consulting cc, 50 Campground Road, Rondebosch, Cape Town, South Africa; P. Dugan, WorldFish Center, P. O. Box 1261 Maadi, Cairo, Egypt; A. S. Halls, Marine Resources Assessment Group Ltd, 47 Princes Gate, London SW7 2QA, UK; J. M. King, Freshwater Ecology Unit, Zoology Department, University of Cape Town, Rondebosch, Cape Town, South Africa; C. V. Minte-Vera, Research Nucleus in Limnology, Ichthyology and Aquaculture (NUPELIA), Department for Biology, State University of Maringa, Brazil; R. E. Tharme, International Water Management Institute (IWMI), P. O. Box 2075, Colombo, Sri Lanka; R. L. Welcomme, Renewable Resources Assessment Group, Faculty of Life Sciences, Department of Environmental Science and Technology, Imperial College of Science Technology and Medicine, Prince Consort Road, London SW7 2BP, UK.

Acknowledgements: We acknowledge the leadership and funding provided by Dr. Patrick Dugan, WorldFish Center, the support provided to all authors by their host organisations, and the Editor of this report, Mahen Chandrasoma, IWMI, for his careful work on the final manuscript. This report has been reviewed prior to publication.

Arthington, A. H.; Baran, E.; Brown, C. A.; Dugan, P.; Halls, A. S.; King, J. M.; Minte-Vera, C. V.; Tharme, R. E.; Welcomme, R. L. 2007. *Water requirements of floodplain rivers and fisheries: Existing decision support tools and pathways for development*. Colombo, Sri Lanka: International Water Management Institute. 74 pp. (Comprehensive Assessment of Water Management in Agriculture Research Report 17)

/ environmental impact assessment / flood plains / fisheries / water requirements / decision support tools / ecology / models / rivers / methodology /

ISSN 1391-9407
ISBN 978-92-9090-656-8

Cover photograph: Sampling fish by cast net during an environmental flow study in the Geum River, South Korea. *Photo Credit:* Angela H. Arthington.

Please send inquiries and comments to: comp.assessment@cgiar.org

Contents

Summary

Fisheries are some of the most valuable natural resources that depend upon natural regimes of river flow for their productivity and full development benefits. Managing rivers to sustain these benefits requires that environmental flow requirements of river fisheries be understood and conveyed effectively into decision-making processes at multiple levels within the river basin. The present research report reviews existing environmental flow methodologies and fisheries production models to determine which combination of existing approaches will provide most potential for development of such decision support tools.

This is the first comprehensive review of the use of environmental flow methodologies for managing large rivers and floodplains for fisheries production. Previous reviews have focused exclusively on the fisheries models themselves and have not explored how these models can be combined with other approaches to understand and predict the impact of changes in river flow regimes on fisheries production. In view of the importance of river fisheries in the livelihoods of the rural poor across much of the tropics, more effective decision-support tools that improve the ecological basis of water management in river basins can play an important part in increasing fisheries productivity and the health and livelihoods of rural populations at the basin scale.

The review concludes that the methodology DRIFT (Downstream Response to Imposed Flow Transformation) combined with use of Bayesian networks and age-structured fisheries models will provide the most promising direction for future research. There will however be cases where this approach will not work and flexibility in the use of other environmental flow methodologies and tools will be required.

The target audience for this review includes river and fisheries scientists concerned with improving water management for the poor of the developing world, and river scientists and managers in general. The results of the research can be best applied through the development of improved environmental flow and fisheries modeling approaches, including studies in the focal basins of the CGIAR Challenge Program on Water and Food.

Water Requirements of Floodplain Rivers and Fisheries: Existing Decision Support Tools and Pathways for Development

A. H. Arthington, E. Baran, C. A. Brown, P. Dugan, A. S. Halls, J. M. King, C. V. Minte-Vera, R. E. Tharme and R. L. Welcomme

Introduction

As demand for freshwater continues to rise and ways are sought to improve water productivity, decision-making bodies at local, basin and national levels require accurate information on the role of river flows in sustaining a wide range of environmental benefits. River and floodplain fisheries are one of these benefits, yielding an estimated 8 million tons of fish and other aquatic natural resources annually, and providing employment, income and an essential supply of animal protein for hundreds of millions of people (FAO statistics). In addition, respect for the diversity of living aquatic flora and fauna is inherent in international conservation instruments including the Conventions on Biological Diversity and Wetlands, which emphasize the duty of signatory nations to preserve the wealth of their living aquatic resources and the aquatic ecosystems that support them. However, as use of water for agriculture, hydropower generation and supplies for domestic and industrial use from river systems has increased, progressive degradation of ecosystem structure and functioning has occurred, with associated declines in fish catches and the disappearance of individual species (Dudgeon et al. 2006; Welcomme et al. 2006). There is now widespread international recognition that future water allocation and flow management in river basins must take account of the needs of river ecosystems, fish and fisheries and the human communities that depend upon them (Bunn and Arthington 2002; Naiman et al. 2002; Postel and Richter 2003; Revenga et al. 2000; Rosenberg et al. 2000).

The World Water Vision, the World Commission on Dams (WCD), the Global Dialogue on Water for Food and Environment, and the Convention on Biological Diversity (CBD), have all responded to these growing pressures upon river ecosystems, calling for a new approach to fisheries and river management as an integral part of efforts to improve the sustainable benefits that people obtain from water. For example, Guidelines 15 and 16 of the WCD call for “Environmental Flow Assessments” and “Maintaining Productive Fisheries” and specify *inter alia* the importance of assessments of the water requirements of fish populations and the mitigation of fish losses on the downstream floodplain through flow releases. Similarly, at the 8th Meeting of the Conference of the Contracting Parties to the Ramsar Convention on Wetlands (online: http://www.ramsar.org/index_cop8.htm), several pertinent resolutions were passed, notably Resolutions VIII.1 (guidelines for the allocation and management of water for maintaining the ecological functions of wetlands), VIII.2 (dealing with the outcomes of the WCD process), and VIII.34 (management of agriculture, wetlands and water resources). These resolutions have recently been superseded by Ramsar Resolution IX.4 (the Ramsar Convention and conservation, production and sustainable use of fisheries resources, i.e., fish, crustaceans, mollusks and algae) passed at the 9th Meeting of the Conference of the Parties to the Convention on Wetlands held in Uganda in 2005 (online: http://www.ramsar.org/res/key_res_ix_index_e.htm). Contracting Parties and

relevant organizations are urged to use the habitat and species conservation provisions of the Convention to support the introduction and/or continuance of management measures that mitigate the environmental impacts of fishing, including the use of spatial management approaches as appropriate, and the Ramsar Secretariat is requested to work with other conventions, instruments and organizations concerned with the conservation of biodiversity and the management of natural resources, including the Food and Agriculture Organization of the United Nations (FAO) at an international and regional level, in order to promote the synergy and alignment of planning and management approaches that benefit the conservation and sustainable management of fisheries resources and recognition of the contribution this makes towards meeting CBD targets and Millennium Development Goals (MDGs).

While this growing international recognition of the importance of riverine resources provides the basis for policy frameworks, efforts to build upon these initiatives and improve water management in individual rivers have been constrained by the lack of specific information on the value of individual fisheries, and the impact of changes in water flow on these resources and the people who depend upon them. At the same time, it is difficult to establish the water management regime required to sustain fisheries and their benefits in the face of increasing water demand (Welcomme et al. 2006). As a result the recent history of river fisheries in the tropics, in particular, has continued to be one of decline and loss of benefits to people and ecosystems (Revenga et al. 2000; Allan et al. 2005).

If the increased awareness and emerging policy framework provided through these international initiatives are to lead to sustained benefits for poor fishing communities, they need to be followed now by the generation of policy relevant information and development of practical tools that can support water management decisions at local, basin and national level. The growing demand for water to service human needs means that increasing quantities will be withdrawn from natural freshwater ecosystems. It is inevitable that this trend to take increasing control of freshwater resources (surface and ground water) on all continents will expand, at least in part, through hard engineering solutions such as dams, channelization and inter-basin water transfer schemes (Naiman et al. 2002). The withdrawal and regulation of water alters the absolute amount available in any one system, the ratio of high to low water volumes and the timing, duration, frequency and rate of change of floods and low flows. The organisms living in aquatic ecosystems have become adapted over long periods of time to the type of flow regime occurring in the river concerned (Arthington et al. 1992; Poff et al. 1997; Richter et al. 1997, among others). Alterations in flow quantity and temporal pattern impact on the species richness, distribution and relative abundance of the organisms present, usually by reducing diversity and densities, as well as forcing shifts in communities towards organisms that are less favored for food and commerce.

These and related issues were reviewed in early 2002 during a workshop on “Managing River Flows to Sustain Tropical Fisheries” held during the International Working Conference on Environmental Flows for River Systems and the Fourth International Ecohydraulics Symposium (Cape Town, South Africa, March 4-8, 2002). This workshop provided participants with an opportunity to examine these issues, and work together to identify practical research priorities and opportunities. Following the workshop, an international collaborative research project was established to review and develop decision-support tools for water allocations that will sustain river fisheries and so improve water productivity at basin level. As the first phase of this project, funded by the Comprehensive Assessment of Water Management in Agriculture (CA) and the WorldFish Center, the present study was undertaken to effect state of the art reviews of the tools currently being used to assess water requirements for river ecosystems and fish/fisheries, and assess their suitability for application in large floodplain rivers, by identifying their strengths and weaknesses, and providing guidance on the further development and testing of improved tools in selected river basins.

Decisions on the allocation of water among users should be made on the basis of the best available information on the sector-by-sector consequences of changing hydrological regimes. The following review examines the various tools that are available to provide information on the impact on river ecosystems and their fisheries (and other natural aquatic resources) of changes in hydrological regime and of differing land-use practices on the river-floodplain system. Some of the methods described directly address the response of fish and fishers to such changes. Others place fisheries in the wider context of other production and biodiversity related issues within the river basin. The methodologies and models described form a complementary suite of tools that is intended to provide planners and policymakers with information as to the consequences for fisheries of various possible changes in hydrological regime resulting from water abstractions and other types of alteration to natural river flow regimes.

Review of Environmental Flow Methodologies

The Conceptual Basis of Environmental Flow Assessment

In many parts of the world there is growing awareness of the pivotal role of the flow regime (hydrology) as a key 'driver' of the ecology of rivers and their associated floodplain wetlands (see Richter et al. 1997; Poff et al. 1997; Bunn and Arthington 2002; Naiman et al. 2002 for reviews). In large part, this recognition has stemmed from an increasing body of scientific knowledge describing the range of negative impacts to riverine ecosystems, and consequent social and economic effects, that are clearly attributable, either directly or indirectly, to alterations of natural flow regimes (as outlined in Rosenberg et al. 2000).

Every river system has a flow regime with particular characteristics relating to flow quantity, temporal attributes such as seasonal pattern of flows, timing, frequency, predictability and duration of extreme events (floods, droughts), rates of change and other aspects of flow variability (Poff et al. 1997; Olden and Poff 2003). Each of these hydrological characteristics has individual (as well as interactive) influences on the physical nature of river channels and habitat structure, biological diversity, recruitment patterns and other key ecological processes sustaining the aquatic ecosystem (Poff et al. 1997; Baron et al. 2002; Bunn and Arthington 2002; Naiman et al. 2002). These processes in turn govern the ecosystem goods and services that rivers provide to humans (e.g., flood attenuation, water purification, fish and fibres, medicines).

Recognition of the escalating hydrological alteration of rivers on a global scale and resultant environmental degradation has led to the gradual establishment of a field of scientific research termed environmental flow assessment (EFA). In simple terms, such an assessment addresses how much of the original discharge of a river should continue to flow down it and onto its floodplains, and when/how often such flows should occur in order to maintain specified, valued features of the ecosystem (Arthington and Pusey 1993; Tharme and King 1998; King et al. 2003; Tharme 2003). An EFA produces one or more descriptions of possible modified hydrological regimes for the river, termed the environmental flow requirement(s) (EFRs) or environmental water allocation(s) (EWAs), each linked to a predetermined objective in terms of the ecosystem's future ecological condition. In

recent literature environmental flows and environmental water allocations refer to the water regime of a river, wetland or coastal zone necessary to maintain the biophysical components, ecological processes and health of aquatic ecosystems, and associated ecological goods and services (Arthington et al. 2006).

Environmental flow assessments are directed at two main types of management response to the impacts of altered flow regimes: either to maintain the hydrological regimes of undeveloped rivers in near to natural condition or at least to offer some degree of protection of river flows and ecosystem characteristics (i.e., river conservation), or, in developed rivers with modified flow regimes, to restore key components and patterns of the original flow regime in the expectation that flow-related ecosystem characteristics will also be restored.

The level of resolution of environmental flow recommendations ranges widely, from a single annual volumetric allocation through to, more commonly nowadays, a comprehensive, modified flow regime where the overall volume of water allocated for environmental purposes is a combination of different monthly and/or event-based flow quantities (Tharme 2003; Arthington et al. 2006). The scale at which assessments are undertaken may also vary widely, for instance, from an entire river basin that includes a regulated mainstream and unregulated tributaries, to a flow restoration project for a single flow-impacted river reach (King et al. 1999). Different methodologies are appropriate over such scales of space and resolution, and in relation to typical project constraints including the time frame for assessment, the availability of data, technical capacity and finances (Tharme 1996; Arthington et al. 2004a, 2004b). As described further below, methods range from rapid, reconnaissance-level approaches for regional, national or basin-wide water resources planning, to resource intensive methodologies for highly exploited river reaches subject to multiple uses of water.

Evolution and Description of the Main Types of Environmental Flow Methods

The origins of environmental flow assessment

Tharme (1996) traced the evolution of environmental flow methods worldwide, observing that historically, the United States of America was at the forefront of research, with the first *ad hoc* methods and a series of more formally documented techniques appearing in the late 1940s and 1970s, respectively. In most other parts of the world, the EFA process became established far later, with approaches to determine environmental water allocations only beginning to appear in the literature in the 1980s. Early on, and still today in some countries, the focus of environmental flow assessment was entirely on the maintenance of economically important freshwater fish species and fisheries in regulated rivers, where a minimum acceptable flow was recommended based almost entirely on predictions of in-stream habitat availability matched against the habitat preferences of one or a few target species. It was assumed that the flows intended to protect target fish populations, habitats and activities would ensure maintenance of other riverine species and communities. Over the past decade or so, a broader, ecosystem approach to EFAs has been adopted in both theory and practice.

Description of methods

Four relatively discrete types of environmental flow methods have become established over time, namely (1) hydrological, (2) hydraulic rating, (3) habitat simulation, and (4) holistic (ecosystem) methods (Table 1).

Hydrological methods

These represent the simplest set of techniques where, at a desktop level, hydrological data, as naturalized, historical monthly or average daily

flow records, are analyzed to derive standard flow indices as recommended flows. Commonly, the EFR is represented by a proportion of flow (often termed the 'minimum flow', e.g., $Q_{95,}$ the flow equalled or exceeded 95% of the time). This low flow quantity is intended to maintain river health, fisheries or other highlighted ecological features at some acceptable level, usually on an annual, seasonal or monthly basis. In a few instances, secondary information in the form of catchment variables, hydraulic, biological and/or geomorphological criteria are also incorporated into the EFR. As a result of their rapidity and lack of ecological refinement, these low resolution hydrological methods are only appropriate at the planning level of water resource development, and in situations where bulk water entitlements must be estimated, or where limited tradeoff of water allocations is required. These volumetric EFRs should be regarded as preliminary flow targets (Tharme 1996; Arthington and Zalucki 1998; Dunbar et al. 1998; Table 1) to be refined at a later stage of water resource planning.

Hydraulic rating methods

These use changes in simple hydraulic variables, such as wetted perimeter or maximum depth, usually measured across single, flow-limited river cross-sections (e.g., riffles), as a surrogate for habitat factors known or assumed to be limiting to target biota. Environmental flows are determined from a plot of the hydraulic variable against discharge, commonly by identifying curve breakpoints where significant percentage reductions in habitat quality occur with decreases in discharge. It is assumed that ensuring some threshold value of the selected hydraulic parameter at altered flows will maintain the in-stream biota and thus, ecosystem integrity. These relatively low resolution techniques are still in use, but have been largely superseded by more advanced habitat modeling tools or assimilated into holistic methodologies (Tharme 1996; Arthington and Zalucki 1998; Table 1).

Habitat simulation or microhabitat modeling methods

These also make use of hydraulic habitat-discharge relationships, but provide more detailed, model-based analyses of both the quantity and suitability of in-stream physical habitat available to target biota under different discharges, on the basis of integrated hydrological, hydraulic and biological data. Flow-related changes in physical microhabitat are modeled in various hydraulic programs, typically using data on depth, velocity, substratum composition and cover and, more recently, complex hydraulic indices (e.g., benthic shear stress) collected at multiple cross-sections within each study river reach. Simulated available habitat is linked with seasonal information on the range of habitat conditions used by target fish or invertebrate species (or life stages, assemblages and/or activities), commonly using habitat suitability index curves. The resultant outputs, in the form of habitat-discharge curves for specific biota, or extended as habitat time and exceedance series, are used to derive optimum environmental flows. The relative strengths and limitations of such methods are described in Tharme (1996) and Arthington and Zalucki (1998), and they are compared with the other types of approach in Table 1.

Holistic (ecosystem) methodologies

As the above three types of methods have evolved in recent times, so has understanding of the broader ecological requirements of fish and other aquatic biota such as invertebrates and aquatic vegetation, and of the interactions between these groups and their physical habitats, as a consequence of altered flow regimes. River ecologists in South Africa and Australia, in particular, have argued for a broader approach to environmental flows and the conservation of river ecosystems (Arthington and Pusey 1993; King and Tharme 1994; Arthington et al. 1998). A conceptual 'holistic' approach to environmental water allocation was first proposed in 1991, based

TABLE 1.
Simplified comparison of the four main types of environmental flow methodology presently used worldwide.

Type	Riverine ecosystem component(s) addressed	Data needs	Expertise	Complexity	Resource intensity	Resolution of output (EFR)	Flexibility	Appropriate level(s) of application
Hydrological	• Whole ecosystem - non-specific	L (primarily desktop) • Virgin/naturalized historical flow records • Some use historical ecological data	L • Hydrological • Some ecological expertise	L	L	L	L	• Reconnaissance level of water resource developments, or as a tool within habitat simulation or holistic methodologies
Hydraulic rating	• Instream habitat for target biota	L-M (desktop, limited field) • Historical flow records • Discharge linked to hydraulic variables - typically single river cross-section • Hydraulic variable(s) as surrogate for habitat-flow needs of target species/assemblages	L-M • Hydrological • Some hydraulic modeling • Some ecological expertise	L-M	L-M	L	L	• Water resource developments where no or limited negotiation is involved, or as a tool within habitat simulation or holistic methodologies
Habitat simulation	• Primarily instream habitat for target biota • Some consider channel form, sediment transport, water quality, riparian vegetation, wildlife, recreation and aesthetics	M-H (desktop and field) • Historical flow records • Many hydraulic variables – multiple cross-sections • Physical habitat suitability data for target species	H • Advanced hydrological modeling • Advanced computer-based hydraulic and habitat modeling • Specialist ecological expertise on physical habitat-flow needs of target species	M-H	H	M-H	M	• Water resource developments, often large-scale, involving rivers of high conservation and/or strategic importance, and/or with complex, negotiated tradeoffs among users, or as method within holistic methodology; primarily developed countries
Holistic	• Whole ecosystem – all/several individual components • In addition to instream and riparian components, some consider: groundwater, wetlands and floodplain, estuary, social dependence on ecosystem and related economic factors	M-H (desktop and field) • Historical flow records • Many hydraulic variables – multiple cross-sections • Biological data on flow- and habitat-related requirements of all biota and ecological components	H • Advanced hydrological modeling • Advanced computer-based hydraulic modeling • Habitat modeling in some cases • Specialist ecological expertise on all individual ecosystem components • Some require social and economic expertise	M-H	M-H	M-H	H	• Water resource developments, typically large-scale, involving rivers of high conservation and/or strategic importance, and/or with complex user tradeoffs; simpler approaches (e.g., expert panel assessments) are appropriate where there are limited tradeoffs among users; developing and developed countries

Source: adapted from King et al. 1999

Notes: EFR – environmental flow requirement

The ratings or ranges in rating, L – low, M – moderate and H – high, pertain to the majority of methodologies within each type. The latest methodologies may be more complex and rate somewhat higher than indicated. Resource intensity is represented in terms of time, cost and technical capacity.

on the natural flow regime as a guide to the environmental conditions that have maintained river ecosystems in their characteristic states. This approach argues that "if the essential features of the natural hydrological regime can be identified and adequately incorporated into a modified flow regime, then, all other things being equal, the extant biota and functional integrity of the ecosystem should be maintained" (Arthington et al. 1992). Likewise, Sparks (1992, 1995) suggested that "rather than optimizing water regimes for one or a few species, a better approach is to try to approximate the natural flow regime that maintained the entire panoply of species."

Importantly, therefore, all holistic methodologies aim to address the water requirements of the entire riverine ecosystem (Arthington et al. 1992) rather than the needs of just a few taxa (e.g., fish species of significance for fishing or conservation, riparian and wetland vegetation, or water birds). They are nowadays underpinned by the "natural flows paradigm" (Poff et al. 1997) and basic principles of river corridor restoration (Ward et al. 2001), and share a common objective - to maintain or restore those characteristics of the natural (or modeled unregulated) flow regime and its variability that are required to maintain or restore the biophysical components and ecological processes of in-stream and groundwater systems, floodplains and downstream receiving waters (e.g., terminal lakes and wetlands, or estuaries).

In any holistic methodology, key flow events are identified in terms of selected criteria defining flow variability, for some or all major components/ attributes of the riverine ecosystem (geomorphology, water quality, vegetation, invertebrates, fish and other vertebrates associated with the aquatic and riparian ecosystem). This analysis is undertaken in either a prescriptive 'bottom-up' or, more recently in a few cases, an interactive 'top-down' process (Arthington et al. 1998) involving structured, workshop-based multi-disciplinary input from a wide range of experts (Tharme and King 1998; King et al. 2000a, 2000b; Arthington et al. 2004a, 2004b). In most approaches to date, a modified flow regime is built from the bottom up by adding flow components to a baseline of zero flow on a flow event or month-by-month timescale, and element-by-element. Each element represents a defined feature of the flow regime intended to achieve particular ecological or other objectives (King and Tharme 1994; Arthington et al. 2004b). In the few 'top-down', scenario-based approaches that exist, environmental flow needs are defined through various levels of change from the natural or other reference flow regime, rendering them less susceptible to any omission of critical flows than bottom-up approaches (Bunn 1998). Top-down approaches address the question - *how much can we modify a river's flow regime before the aquatic ecosystem begins to noticeably change or becomes seriously degraded?* Top-down approaches appear to be relatively rare, the two prominent ones being Downstream Response to Imposed Flow Transformation or DRIFT (King et al. 2003) and the Benchmarking Methodology (Brizga et al. 2002; Arthington et al. 2004b): for discussion of similarities and differences, see Arthington et al. (2004b, 2006).

The most advanced holistic methodologies routinely utilize several of the tools for hydrological, hydraulic and physical habitat analysis featured in the other methods discussed above. Moreover, they may also rely on flow-ecology models to be sufficiently predictive in their outputs (Tharme and King 1998; Bunn and Arthington 2002). Further discussion of holistic methodologies can be found in section: *Holistic Methodologies and the Relevance of Hydrology-Ecology Models,* and a summary comparison of this approach with hydrological and habitat methods is provided in Table 1.

Other approaches

In addition to the four main categories of methodology, Tharme (1996, 2003), Dunbar et al. (1998) and Arthington and Pusey (2003) recognize an array of approaches that incorporate characteristics of more than one of the four types, including partially holistic environmental flow methods (EFMs), referred to as 'combination' approaches (Tharme 2003). There are also a

number of techniques not designed with environmental flow assessment in mind, but which have been variously adapted for this purpose (termed 'other' approaches). Together, these approaches contribute to the diversity and total number of environmental flow methods and tools available globally (see section: *An Overview of Global Trends in Method Development and Application).*

An Overview of Global Trends in Method Development and Application

Several comprehensive international reviews of environmental flow methods have been published over the past decade (*inter alia,* Arthington and Pusey 1993; Growns and Kotlash 1994; Tharme 1996; Jowett 1997; Stewardson and Gippel 1997; Dunbar et al. 1998; Arthington 1998; Arthington et al. 1998; Arthington and Zalucki 1998; King et al. 1999; Tharme 2003; Arthington et al. 2004b). Further information on new world developments in environmental flow methodologies can be found in papers of the unpublished proceedings of the joint South African Environmental Flows for River Systems Working Conference and Fourth International Ecohydraulics Symposium (held in Cape Town, March 2002). Some papers from this conference are available on CD, while others have been published in *River Research and Applications* (Vol. 19, 2003).

Building on these and other sources, an analysis by Tharme (2003) of the numbers of environmental flow methods developed for rivers (and their associated floodplain wetlands) within the categories described above, revealed some 207 discrete environmental flow techniques applied in 44 countries (within North America, Central and South America, Europe and the Middle East, Africa, Australasia, and the rest of Asia). A further seven countries demonstrated some evidence of early environmental flow initiatives with further advances as yet unreported.

Hydrological methods are the most frequently used approach for environmental flow assessment, at 30 percent of the global total (Tharme 2003). At least 61 different hydrological indices or techniques have been applied, most commonly the Tennant or Montana Method (Tennant 1976) and, even more simplistically, various percentages of average annual flow. Habitat simulation methods are second only to hydrological EFMs worldwide, representing 28 percent of all methods. Within the approximately 58 approaches documented, a subset of microhabitat modeling approaches of similar character and data requirements represents the current state-of-the-art. Of the latter group, the Instream Flow Incremental Methodology, IFIM (including the Physical Habitat Simulation Model, PHABSIM; Bovee 1982; Milhous et al. 1989; Stalnaker et al. 1994), is currently the most commonly used environmental flow method globally, with applications in over 20 countries (Tharme 2003). Although it has been variously adapted over the years, it remains focused primarily on the in-stream flow requirements of target fish and, to a lesser extent, invertebrates. Applications, strengths and limitations of IFIM/PHABSIM are well documented (e.g., Arthington and Pusey 1993; King and Tharme 1994; Tharme 1996; Pusey 1998 and the critiques cited therein).

Of 23 hydraulic rating methods reported in use (11% of the global total), only the generic Wetted Perimeter Method remains in wide use today (Gippel and Stewardson 1998). A fairly high number of combination type methodologies (17% overall) exist, the most common being the Managed Flood Release Approach of Acreman et al. (2000) or similar methods based on experimental flow releases. The former approach has been employed in river floodplain restoration projects in Africa, for fisheries production and sustaining dependent livelihoods, but is only partially holistic in nature. Holistic methodologies currently represent around 8 percent of the global total (Tharme 2003), and the individual methodologies comprising this total are described

at length in sections: *Holistic Methodologies and the Relevance of Hydrology-Ecology Models and The DRIFT Methodology Further Examined* below. Finally, the smallest global proportion of methodologies (6.8%) represents alternative ('other') approaches, many of which are based on multivariate regression analysis.

Applications of environmental flow methods are addressed within a hierarchical framework in many countries (King et al. 1999), at two or more stages: (1) reconnaissance-level assessment, primarily using hydrological methods; and (2) comprehensive assessment, using either habitat simulation or holistic methodologies; examples are provided in Arthington et al. (2003, 2004b), King et al. (2003) and Tharme (2003). In developed countries of the Northern Hemisphere particularly, habitat simulation methods remain the most commonly applied tools at levels of assessment beyond planning estimates, with recent efforts concentrated on advances in multi-dimensional habitat modeling. Holistic methodologies, predominantly used in South Africa and Australia, have begun to attract growing international interest in both developed and developing regions of the world, with strong expressions of interest by some 12 countries in Europe, Latin America, Asia and Africa (Tharme 2003). King et al. (1999) and Tharme (2003) consider such methodologies to be especially appropriate in developing countries, due to the need for resource protection at an ecosystem scale and the direct dependence of local people on the goods and services provided by aquatic ecosystems for food and broader livelihood security.

Importantly, although the analysis of global trends conducted by Tharme (2003) showed a recent, marked increase in the number of countries undertaking environmental flow assessment, there remain a number of countries that have not yet embodied these concepts in water resources policy and legislation, and where the role of environmental flows in the long-term maintenance and sustainability of aquatic resources is yet to be fully recognized. Significantly, while just over half of the countries representing the developed world are reported to be routinely involved in environmental flow initiatives, at various degrees of advancement, only 11 percent of developing countries are active in any such research to date (Tharme 2003).

Holistic Methodologies and the Relevance of Hydrology-Ecology Models

Overview

At least 16 methodologies based on the holistic principles described in section: *Review of Environmental Flow Methodologies* have been developed over the last ten years (Tharme 2003). The collaboration among Australian and South African researchers that resulted in the establishment of the first conceptual holistic framework for environmental flow assessment (see section: *Description of Methods*) provided the direction and momentum for both countries to develop, in parallel and often in collaboration, most available methodologies of this type (Tharme 1996).

In South Africa, the Building Block Methodology (BBM)(King and Tharme 1994; Tharme and King 1998; King and Louw 1998; King et al. 2000a), has advanced through numerous applications linked to water resource projects to become the most commonly applied holistic methodology worldwide (Tharme 2003). It is used especially in the modified forms legally required for intermediate and comprehensive determinations of the Ecological Reserve (and recently incorporating the Flow Stress-Response

Method, O'Keeffe et al. 2002). Other holistic methodologies applied locally include Downstream Response to Imposed Flow Transformation (DRIFT, King et al. 2003), detailed in section: *The DRIFT Methodology Further Examined* and a focal tool in the present overview, and the Environmental Flow Management Plan Method (Muller 1997; DWAF 1999). Tharme (1996, 2003 tracks the establishment of holistic methodologies within the broader context of environmental flow assessment in South Africa.

Australia's progression from hydrological 'rules-of-thumb' to the present-day emphasis on the water requirements of river ecosystems has been outlined in several review papers (Arthington and Pusey 1993, 2003; Cottingham et al. 2002; Arthington et al. 2004b, 2006). Other recent developments are presented in a special issue of the *Australian Journal of Water Resources* (Volume 5, No. 1, 2002). Australian holistic methodologies are numerous and diverse in character, and include the original conceptual Holistic Approach (Arthington et al. 1992; Davies et al. 1996; Arthington 1998; Pettit et al. 2001), Expert Panel Assessment Method (EPAM; Swales and Harris 1995), Scientific Panel Assessment Method (SPAM; Thoms et al. 1996), Habitat Analysis Method (HAM, Walter et al. 1994; Burgess and Vanderbyl 1996), Flow Restoration Methodology (Arthington et al. 2000), Benchmarking Methodology (Brizga 2000; Brizga et al. 2002), Flow Events Method (Stewardson 2001), and FLOWS (a method used routinely in Victoria).

Additional holistic methodologies developed and applied elsewhere include the River Babingley Method (Petts et al. 1999) developed in England, and the Adapted BBM-DRIFT Methodology developed in Zimbabwe (Steward et al. 2002).

Brief notes on several of these approaches follow in section: *Comparative Description of Methodologies* (see also Arthington et al. 2004b).

Comparative Description Of Methodologies

For comparative purposes, selected holistic methodologies from the broader group are summarized in Table 2, in terms of their origins, key features, strengths, limitations, and present status (adapted from Tharme 2003). Further details of these various methodologies are available in the source references provided in Table 2, as well as in the review papers listed previously.

Essential features of holistic methodologies

All holistic methodologies must address two essential questions. First, which flows are important, and linked to this question, what quantities and temporal patterns of flow are needed to sustain river biota and ecosystems? Second, what are the potential impacts of altered flows on the aquatic ecosystem? Both of these related questions require understanding, and preferably, quantification of the various relationships between hydrology, geomorphology, water quality and ecological systems.

Which flows are important?

Six major categories of flow characteristics have been identified as being of particular geomorphological and ecological relevance, as well as being sensitive to the changes produced in flow regimes by impoundment, diversions, groundwater exploitation, use of water for hydropower generation and catchment land-use changes (Arthington et al. 1998; Richter et al. 1997). Important flow characteristics are (Poff et al. 1997):

TABLE 2.
Summary of a range of holistic environmental flow methodologies, presented in no particular order, highlighting salient features, strengths and limitations, as well as their current status in terms of development and application.

Methodology (Key references)	Origins	Main features, strengths and limitations	Status
Benchmarking Methodology (DNR 1998b, cited in Arthington 1998; Brizga 2000; Brizga et al. 2002)	Developed in Queensland, Australia, by local researchers and Department of Natural Resources (DNR), to provide a framework for assessing risk of environmental impacts due to water resource development, at basin scale	Comprehensive, scenario-based, top-down approach for application at basin scale, using field and desktop data for multiple river sites; EFM has 4 main stages – (1) establishment: formation of multidisciplinary expert panel (TAP) and development of hydrological model, (2) ecological condition and trend assessment: development of spatial reference framework, assessment of ecological condition for suite of ecosystem components (using 5-point rating of degree of change from reference condition and appropriate methods for each component), development of generic models (conceptual, empirical) defining links between flow regime components and ecological processes, selection of key flow indicators/statistics with relevance to these relationships, modeling-based assessment of hydrological impacts, (3) development of risk assessment of potential framework to guide evaluation impacts of future water resource development/ management scenarios: benchmark models are developed for all/some key flow indicators showing levels of risk of ecological/geomorphological impacts associated with different degrees of flow regime change, risk levels are defined by association with benchmark sites which have undergone different degrees of flow-related change in condition, link models are used to show how the modeled flow indicators affect ecological condition, (4) evaluation of future WRD scenarios, using risk assessment and link models, ecological implications of scenarios and associated levels of risk readily expressed in graphical form; EFM is particularly suited to data poor situations; potential for use in developing countries context and for application to other aquatic ecosystems; utilizes a wide range of specialist expertise; presents a comprehensive benchmarking process; provides several ways of developing risk assessment models, guidance on key criteria for assessing condition, and key hydrological and performance indicators; recent approach built on several preceding EFA initiatives; no explicit consideration of social component, but with scope for inclusion (note that socio-economic issues are evaluated separately by DNR and considered when the final EF recommendations are being decided); requires evaluation of several aspects (e.g., applicability/ sensitivity of key flow statistics, degree to which benchmarks from other basins/sites are valid considering differences in river hydrology and biota); recommends a monitoring programme and additional research on important issues, as crucial components of EF implementation.	Sole holistic EFM for basin-scale assessment and assessing risk of environmental impacts due to water resource development; adopted for routine application in Queensland with applications in 8 basins; under consideration for use in Western Australia; only applied in Australia to date
Holistic Approach (Arthington et al. 1992; Davies et al. 1996; Arthington 1998)	Developed in Australia to address EFRs of entire riverine ecosystem; shared conceptual basis with BBM, and the theoretical and conceptual basis of the Benchmarking Methodology	Loosely structured set of methods for bottom-up construction of EF regime, with no explicit output format; principally represents a flexible conceptual framework, elements of which have been adapted in a variety of ways into several Australian holistic methodologies and for individual studies; lack of structured set of procedures and clear identity for EFM hinders rigorous routine application; basic tenets and assumptions as per BBM, which was derived from it; systematic construction of a modified flow regime, on a month-by-month and flow element-by-element basis, to achieve predetermined objectives for future river condition; incorporates more detailed assessment of flow variability than early BBM studies; includes method for generating tradeoff curves for examining alternative water use scenarios; some risk of inadvertent omission of critical flow events; represents the theoretical basis for most other holistic EFMs; recommends a monitoring programme as a crucial component of holistic flow assessments.	Represents conceptual basis of most other holistic EFMs; applied in various forms in Australia

(Continued)

TABLE 2. (Continued)
Summary of a range of holistic environmental flow methodologies, presented in no particular order, highlighting salient features, strengths and limitations, as well as their current status in terms of development and application.

Methodology (Key references)	Origins	Main features, strengths and limitations	Status
Building Block Methodology (BBM) (King and Louw 1998; King et al. 2000)	Developed in South Africa by local researchers and DWAF, through application in numerous water resource development projects to address EFRs for entire riverine ecosystems under conditions of variable resources; adapted for intermediate and comprehensive determinations of the ecological Reserve under the new SA Water Law	Rigorous and extensively documented (manual available); prescriptive bottom-up approach with interactive scenario development; moderate to highly resource intensive; developed to differing extents for both intermediate-level (2 months) or comprehensive (1-2 years) EFAs, within South Africa's Reserve framework; based on a number of sites within representative/critical river reaches; includes a well established social component (dependent livelihoods); functions in data poor/rich situations; comprises 3-phase approach - (1) preparation for workshop, including stakeholder consultation, desktop and field studies for site selection, geomorphological reach analysis, river habitat integrity and social surveys, objectives setting for future river condition, assessment of river importance and ecological condition, hydrological and hydraulic analyses, (2) multidisciplinary workshop-based construction of modified flow regime through identification of ecologically essential flow features on a month-by-month (or shorter time scale), flow element-by-flow element basis, for maintenance and drought years, based on best available scientific data, (3) linking of EFR with water resource development engineering phase, through scenario modeling and hydrological yield analysis; EFM exhibits limited potential for examination of alternative scenarios relative to DRIFT, as BBM EF regime is designed to achieve a specific predefined river condition; incorporates a monitoring programme and additional research on important issues, inadvertent omission of critical flow events; high potential for application to other aquatic ecosystems; links to external stakeholder/public participation processes; flexible and amenable to simplification for more rapid assessments; less time, cost and resource intensive than DRIFT; shared conceptual basis with Holistic Approach; applicable to regulated/unregulated rivers, and in flow restoration context; Flow Stressor-Response Method facilitates a top-down, scenario-based assessment of alternative flow regimes, each with expression of the potential risk of change in river ecological condition.	Most frequently used holistic EFM globally, applied in 3 countries; adopted as the standard EFM for South African Reserve determinations

(Continued)

TABLE 2. (Continued)
Summary of a range of holistic environmental flow methodologies, presented in no particular order, highlighting salient features, strengths and limitations, as well as their current status in terms of development and application.

Methodology (Key references)	Origins	Main features, strengths and limitations	Status
Downstream Response to Imposed Flow Transformation (DRIFT) Process (King et al. 2003)	Developed in southern Africa by Southern Waters and Metsi Consultants (with inputs from Australian and southern African researchers) to address need for interactive scenario-based holistic EFM with explicit socio-economic component	Appropriate for comprehensive EFAs (1-3 years) based on several sites within representative/critical river reaches; flexible, interactive, top-down, scenario-based process comprised of 4 modules - (1) biophysical module: used to describe present ecosystem condition, to predict how it will change under a range of different flow alterations for synthesis in a database, uses generic lists of links to flow and relevance for each specialist component, direction scenario and severity of change are recorded to quantify each flow-related impact, (2) sociological module: used to identify subsistence users at risk from flow alterations and to quantify their links with the river in terms of natural resource use and health profiles, (3) scenario development module: links first 2 modules through querying of database, to extract predicted consequences of altered flows (with potential for presentation at several levels of resolution) used to create flow scenarios (typically 4 or 5), (4) economic module: generates description of costs of mitigation and compensation for each scenario; EFM modules require refinement; well developed ability to address socio-economic links to ecosystem; considerable scope for comparative evaluation of alternative modified flow regimes; resource intensive, high potential for application to other aquatic ecosystems; amenable to simplification for more rapid assessments; well documented; uses many successful features of other holistic EFMs; same conceptual basis as BBM and Holistic Approach; exhibits parallels with benchmarking approaches; output is more suitable for negotiation of tradeoffs than in BBM/other bottom-up approaches, as implications of not meeting the EFR are readily accessible; links to external public participation process and macro-economic assessment; generic lists provide clear parameters for inclusion in a monitoring programme; approach provides limited consideration of synergistic interactions among different flow events; limited inclusion of flow indices describing system variability; applicable to regulated/unregulated rivers and for flow restoration; recommends a monitoring programme and additional research on important issues, as crucial components of EF implementation.	EFM with most developed capabilities for scenario analysis and explicit consideration of social and economic effects of changing river condition on subsistence users; limited application to date, within southern Africa

(Continued)

TABLE 2. (Continued)

Summary of a range of holistic environmental flow methodologies, presented in no particular order, highlighting salient features, strengths and limitations, as well as their current status in terms of development and application.

Methodology (Key references)	Origins	Main features, strengths and limitations	Status
Adapted BBM-DRIFT Methodology (Steward et al. 2002)	Developed in Zimbabwe by Mott MacDonald Ltd. in collaboration with Zimbabwe National Water Authority (with input from South Africa) through adaptation of key elements of BBM and DRIFT, in response to requirements in new Water Act for EFAs	Simplified top-down, multidisciplinary team approach, for use in highly resource-limited (including data) situations and with direct dependencies by rural people on riverine ecosystems; combines pre-workshop data collection phase of BBM with DRIFT's scenario-based workshop process; comprises 3 phases - (1) preparation for workshop as per BBM/DRIFT, but excluding certain components (e.g., habitat integrity and geomorphological reach analyses) and with limited field data collection, (2) workshop, with simplified DRIFT process linking the main geomorphological, ecological and social impacts with elements of the flow regime (based on assessments of impact and severity for component-specific generic lists), used to construct a matrix, (3) use of matrix in evaluating development options, where the matrix indicates ecosystem aspects that are especially vulnerable/important to rural livelihoods, socially and ecologically critical elements of the flow regime and EF recommendations for mitigation; EFM incorporates more limited ecological and geomorphological assessments than BBM/DRIFT; limited coverage of key specialist disciplines; no link to system for defining target river condition; limited capability for scenario development; especially appropriate in developing countries context; requires further development and validation; would benefit from inclusion of economic data.	Under early development, single documented application to date
Flow Events Method (FEM) (Stewardson 2001)	Developed by Australian Cooperative Research Centre for Catchment Hydrology to provide state agencies with a standard approach for EFAs	Top-down method for regulated rivers; considers the maximum change in river hydrology from natural for key ecologically relevant flow events, based on empirical data or expert judgement; considered a method of integrating existing analytical techniques and expert opinion to identify important aspects of the flow regime; EFM comprises 4 steps – (1) identification of ecological processes (hydraulic, geomorphic and ecological) affected by flow variations at range of spatial and temporal scales, (2) characterization of flow events (e.g., duration, magnitude) using hydraulic and hydrological analyses, (3) description of the sequence of flow events for particular processes, using a frequency analysis to derive event recurrence intervals for a range of event magnitudes, (4) setting of EF targets, by minimizing changes in event recurrence intervals from natural/reference or to satisfy some constraint (e.g., maximum % permissible change in recurrence interval for any given event magnitude); EFM's singular development appears to be analysis of changes in event recurrence intervals with altered flow regimes; draws greatly on established procedures of other complex EFMs (e.g., BBM, FLOWRESM, and DRIFT); may be used to (1) assess the ecological impact of changes in flow regimes, (2) specify EF management rules/targets, (3) optimize flow management rules to maximize ecological benefits within constraints of existing WRD schemes; possibly places undue emphasis on frequency compared with other event characteristics; no social component; requires additional validation; incorporated within various expert-panel/other assessment frameworks; viewed by several researchers as a tool for use in various holistic EFMs rather than as an EFM in its own right.	Recent approach with few applications in Australia to date; often linked to expert-panel approaches

(Continued)

TABLE 2. (Continued)
Summary of a range of holistic environmental flow methodologies, presented in no particular order, highlighting salient features, strengths and limitations, as well as their current status in terms of development and application.

Methodology (Key references)	Origins	Main features, strengths and limitations	Status
Flow Restoration Methodology (FLOWRESM) (Arthington et al. 2000)	Developed in a study of the Brisbane River, Queensland, Australia, for specifically addressing EFRs in river systems exhibiting a long history of flow regulation and requiring flow restoration	Primarily bottom-up, field and desktop approach appropriate for comprehensive (or intermediate) EFAs; designed for use in intensively regulated rivers with emphasis on identification of the essential that need to be built back into the hydrological regime to shift the regulated river system towards the pre-regulation state; EFM uses an 11-step process in 2 stages, in which the following are achieved - (1) review of changes to the river hydrological regime (focusing on unregulated, present day and future demand scenarios), (2) series of 8 steps within scenario-based workshop, using extensive multidisciplinary specialist input: determination of flow-related environmental effects for low and high flow months, rationale and potential for restoration of various flow components so as to restore ecological functions, and establishment of EFRs based on identification of critical flow thresholds/ flow bands that meet specified ecological/other objectives, (3) develop series of EF scenarios and assess implications of multiple scenarios for system yield, (4) outline remedial actions not related to flow regulation - alternatives to flow restoration (e.g., physical habitat restoration) are evaluated when some elements of pre-regulation flow regime cannot be restored fully for practical/legal reasons, (5) outline monitoring strategy to assess benefits of EFRs; EFM represents hybrid of Holistic Approach and BBM; particular relevance to rivers regulated by large dams, but applicable to any river system regulated by infrastructure or surface/groundwater abstraction; includes well developed hydrological and ecological modeling tools; more rigorous than expert-panel methods; some risk of inadvertent omission of critical flow events; includes flexible top-down process for assessing ecological implications of alternative modified flow regimes; potential for adoption of full benchmarking process to rank outcomes of not restoring critical flows; requires documentation of generic procedure for wider application.	Most comprehensive EFM for flow-related river restoration; single application in Australia to date; however, EFM report is being used as a procedural guide in other recent EF applications (e.g., Ord River study, Western Australia)
River Babingley (Wissey) Method (Petts et al. 1999)	First developed for application in groundwater-dominated rivers, Anglian Region of England	Bottom-up field and desktop approach; EAFR (EF regime) defined in 4 stages - (1) ecological assessment of river and specification of an ecological objective comprising specific targets (for river components and biota), (2) determination of 4 general and 2 flood benchmark flows to meet the specified targets, (3) use of flows to construct 'ecologically acceptable hydrographs', which may include provision for wet years and drought conditions, (4) assignment of acceptable flow frequencies and durations to the hydrographs, and their synthesis into a flow duration curve, the EAFR; EFM uses hydro-ecological models, habitat and hydrological simulation tools to assist in identification of benchmark flows and overall EAFR; allows for flexible examination of alternative EF scenarios; loosely structured approach, with limited explanation of procedures for integration of multidisciplinary input; risk of omission of critical flow events from EAFR; specific to baseflow-dominated rivers and requires further research for use in flashy catchments.	Relatively limited application to date; general approach appears to have been extended to other EFA studies in the United Kingdom

(Continued)

TABLE 2. (Continued)
Summary of a range of holistic environmental flow methodologies, presented in no particular order, highlighting salient features, strengths and limitations, as well as their current status in terms of development and application.

Methodology (Key references)	Origins	Main features, strengths and limitations	Status
Habitat Analysis Method (Walter et al. 1994; Burgess and Vanderbyl 1996; Arthington 1998)	Developed by former Queensland Department of Primary Industries, Water Resources (now DNR), Australia, as part of water allocation and management planning initiative	Relatively rapid, inexpensive, basin-wide reconnaissance method for determining preliminary EFRs at multiple points in catchment (rather than at a few critical sites); superior to simple hydrological EFMs, but inadequate for comprehensive EFAs; field data limited/absent; bottom-up process of 4 stages using TAP - (1) identification of generic aquatic habitat types existing within the catchment, (2) determination of flow-related ecological requirements of each habitat (as surrogate for EFRs for aquatic biota), using small group of key flow statistics, plus select 'biological trigger' flows and floods for maintenance of ecological/geomorphological processes, (3) development of bypass flow strategies to meet EFRs, (4) development of EFR monitoring strategy; EFM represents an extension of expert panel approaches (EPAM, SPAM), with conceptual basis and assumptions adapted from Holistic Approach; little consideration of specific flow needs of individual ecological components; requires standardization of process, refinement of flow bands linked to habitats and addition of flow events related to needs of biota; represents a simplified version of the Holistic Approach; largely superseded by Benchmarking Methodology.	Precursor of Benchmarking Methodology within WAMP initiatives; several applications within Australia
Environmental Flow Management Plan Method (FMP) (Muller 1997; DWAF 1999)	Developed in South Africa by the Institute for Water Research, for use for intensively regulated river systems	Simplified bottom-up approach, applicable in highly regulated and managed systems with considerable operational limitations; considered for use within South Africa Reserve determination process only where BBM or equivalent approach cannot be followed; workshop-based, multidisciplinary assessment including ecologists and system operators; 3-step process - (1) definition of operable reaches for study river and site selection, establishment of current operating rules, (2) determination of current ecological status and desired future state, (3) identification of EFRs using similar procedures to BBM; EFM has limited scope for application; structure and procedures for application are not formalized or well documented; poorly established post-workshop scenario phase; no evaluation undertaken; considerably more limited approach than FLOWRESM.	Limited to 3 applications; only used in South Africa to date uncertain status within the national Reserve framework
Expert Panel Assessment Method (EPAM) (Swales and Harris 1995)	First multidisciplinary panel based EFM used in Australia, developed jointly by the New South Wales departments of Fisheries and Water Resources	Bottom-up, reconnaissance-level approach for initial assessment of proposed WRDs; rapid and inexpensive, with limited field data collection; site-specific focus; applicable primarily for sites where dam releases are possible; relies on field-based ecological interpretation, by a panel of experts, of different multiple trial flow releases (ranked in terms of scored ecological suitability) from dams, at one/few sites, to determine EFR (typically as flow percentiles); low resource intensity; limited resolution of EF output; aims to address river ecosystem health (using fish communities as indicators), rather than to assess multiple ecosystem components; based on same concepts as Holistic Approach and BBM; strongly reliant on professional judgement; limited subset of expertise represented by panel (e.g., fish, invertebrates, geomorphology); simplistic in terms of the range of ecological criteria and components assessed (but scope for inclusion of additional ones) and the focus on fish; no explicit guidelines for application; poor congruence in opinion of different panel members (e.g., due to subjective scoring approach, individual bias); requires further validation; led to development of more advanced, but similar SPAM, Snowy Inquiry Methodology and other expert panel approaches.	Applied only in Australia; several applications, both in original and variously modified forms

(Continued)

TABLE 2. (Continued)
Summary of a range of holistic environmental flow methodologies, presented in no particular order, highlighting salient features, strengths and limitations, as well as their current status in terms of development and application.

Methodology (Key references)	Origins	Main features, strengths and limitations	Status
Scientific Panel Assessment Method (SPAM) (Thoms et al. 1996; Cottingham et al. 2002)	Developed during an EFA for the Barwon-Darling River System, Australia	Bottom-up field (multiple sites) and desktop approach appropriate for provision of interim or intermediate level EFAs; evolved from EPAM as more sophisticated and transparent expert-panel approach; aims to determine a modified flow regime that will maintain ecosystem health; differs from EPAM in that key features of the ecosystem and hydrological regime and their interactions at multiple sites are used as basis for EFA; EFR process includes - (1) identification of management performance criteria by panel of experts for 5 main ecosystem components: fish, trees, macrophytes, invertebrates and geomorphology, (2) application of the criteria for three elements (and associated descriptors) identified as exerting an influence on the ecosystem components (viz. flow regime, hydrograph and physical structure at 3 spatial scales), (3) workshop-based cross-tabulation approach to identify and document generalized responses/impacts for each ecosystem components to each specific descriptor (for each element), so as to relate flow regime attributes to ecosystem responses and EFRs; incorporates system hydrological variability and elements of ecosystem functioning; includes stakeholder-panel member workshop for EFR refinement; many conceptual features and methodological procedures in common with the Holistic Approach and BBM; well-defined EFA objectives; some potential for inclusion of other ecosystem components; led to the evolution of other expert-panel approaches; limited use of field data; poor definition of output format for EFR; moderately rapid, flexible and resource-intensive; simpler, less quantitative supporting evidence, and less rigorous than FLOWRESM, BBM and DRIFT; recent applications and limitations reviewed and need for a Best Practice Framework identified.	Appears limited to a single application in Australia in its original form; general approach variously modified for other expert-panel based EFAs

Source: adapted from Tharme 2003

Notes: EF - environmental flow

EFA - environmental flow assessment

EFR - environmental flow requirement

EFM - environmental flow methodology

Further information on the strengths and deficiencies of individual holistic methodologies is provided in Tharme (1996) and Arthington et al. (2004b).

- the magnitude of river flows at any given time,
- the timing of occurrence of particular flow conditions,
- the frequency of occurrence of particular flows such as flood flows,
- the duration of time over which specific flow conditions extend,
- the rate of change in flow conditions such as rise and fall of flood waters, and
- the seasonality, variability and predictability of the overall flow regime

As expressed recently (Olden and Poff 2003), these features of river flow regimes must be "retained or restored to maintain aquatic biodiversity, ecosystem processes and the long-term evolutionary potential of freshwater ecosystems." It is also well recognized that these flows also sustain the ecological goods and services provided by rivers to society (King et al. 2003; Naiman et al. 2002).

Holistic environmental flow methods tend to apply relatively comprehensive sets of flow statistics or 'indicators' that address many of these discharge characteristics. For example, Australian benchmarking studies typically use statistical descriptors of total annual flow, inter-annual variability, seasonal variability, zero flow, low to medium flow, various categories of high flows/floods, and rates of hydrograph rise and fall (Brizga et al. 2002). Other environmental flow studies have specified flows in relation to a range of 'functional' flow levels or 'bands' relevant to particular geomorphological and ecological attributes and processes (e.g., Arthington et al. 2000), usually including groundwater, baseflow, low flow, mid flow, high flow, bankful flow, overbank flows and catastrophic flows (e.g., SKM 2001). The DRIFT Methodology considers different categories of flow (wet and dry season low flows; small, medium and large floods) that maintain different parts of the river ecosystem (see Arthington et al. 2003; King et al. 2003).

Increasingly, holistic environmental flow studies have incorporated qualitative and quantitative models to define the relationships between river hydrology, geomorphology, water quality and ecological structure/functioning. Qualitative model types are outlined in section: *Qualitative Hydrology-Ecology Models.* These models are not predictive but can be used to develop testable hypotheses linking flow and flow-related variables to biological or ecological response variables over time. Quantitative models are reviewed in section: *River Fisheries Models.*

Qualitative hydrology-ecology models

Six groups of qualitative models can be recognized in the literature and reports describing environmental flow studies: pictorial explanatory models, link models, gradient models, flow chart models, time series models and conceptual ecosystem models.

Pictorial explanatory models

Pictorial explanatory models have been applied widely in Australia to depict the impacts of various degrading processes including flow regulation on aquatic ecosystems, and are also used as a communication tool to explain the results of river condition or 'health' assessments. Pictorial models are used in the Benchmarking Methodology to integrate information relating to the influence of flow on various river ecosystem components (Brizga et al. 2002). Information can be presented in a range of pictorial formats, such as whole of river system, longitudinal profile, or river cross-sections. These models summarize the current state of knowledge of flow-ecosystem relationships and are supported by references to the literature or personal observations.

Link models

Link models show how environmental factors (such as components of the flow regime) and various attributes of aquatic ecosystems relate to one another and interact. These models are used to show the relevance and function of flow indicators, and interrelationships between various ecosystem components. Link models do not explain the processes involved in the interrelationships but they may provide information on the direction of change. For example, a classic link model in the geomorphological literature is the set of relationships published by Schumm (1969) between changes in flow and bedload (increase or decrease in either or both of these parameters), and measures of channel morphology (including width, depth, width/depth ratio, meander wavelength and sinuosity).

Link models showing primary linkages of five groups of key flow indicators to non-tidal rivers and estuaries have become part of the environmental flow assessment framework used in the Benchmarking Methodology (Brizga 2000; Brizga et al. 2002).

Gradient models

Gradient models show the effect of gradients of variables on an ecosystem component. Locations along environmental gradients can be used to determine likely ecological responses to a particular environmental variable, such as discharge. Vegetation frequently exhibits a predictable zonation pattern about stream banks in terms of structure (life form representation, height and stratification) and floristics (species presence/absence and relative abundance). A simple gradient model can be developed to demonstrate the interaction between vegetation structure in terms of occurrence of vegetation types/predominant functional guilds, changes in moisture availability and flood disturbance within the landscape. The BBM, Flow Restoration, DRIFT and Benchmarking methodologies all use the relationships between river discharge, moisture levels and gradients of riparian vegetation to establish environmental flow recommendations.

Flow chart models

Flow chart models show the processes underlying relationships between flow regime change and geomorphological and ecological responses. Flow chart models expand on specific links developed in link models. They have been applied in benchmarking studies to underpin assessments of the condition of plant and fish assemblages in relation to flow regime change, for example.

Time series models

Time series models show the response of geomorphological and/or ecological variables to flow regime change through time. Models may refer to a range of timescales: e.g., seasonal, annual, long-term. Their utility lies in presenting processes with a temporal component in a simple diagrammatic format.

Conceptual ecosystem models

There are a number of ecological models describing ecosystem processes in relation to environmental factors, including the River Continuum Concept (Vannote et al. 1980), the Flood-Pulse Concept (Junk et al. 1989) and the Riverine Productivity Model (Thorp and DeLong 1994). Each model differs considerably in its emphasis on upstream and local in-stream *versus* floodplain sources of organic matter. The Serial Discontinuity Concept (Ward and Stanford 1983) describes how rivers might be expected to respond to the construction of a large dam at various points along the river continuum. It is based upon the River Continuum Concept and as such does not take the alternative models of ecosystem structure into account.

Implications of the Various Holistic Methodologies

Of the global suite of environmental flow methods currently available, the majority, in the form of hydrology-based approaches, have been applied at a desktop planning level, with little, if any, ecological input. For the reasons outlined in Table 3, such methods are totally inadequate for the detailed, flexible kinds of assessment of environmental flows required for large rivers with highly diverse biological assemblages and major fisheries. At best they can provide only approximate volumes of water to be allocated to the aquatic ecosystem, or single minimum flows, thereby precluding any scenario-based tradeoff analyses of basin water use. Although the most recent and sophisticated hydrological methodologies, such as the Range of Variability Approach (RVA) of Richter et al. (1996; 1997) more adequately address the full range of flow variability in rivers, and are purported to be more ecologically relevant, they still do not address the flow needs of the whole riverine ecosystem or specific biotic components. RVA is perhaps most useful for providing interim environmental flow recommendations whilst more detailed analyses are in progress (Richter et al. 1996, 1997).

Of all habitat simulation methods, the state-of-the-art tools such as IFIM provide for more comprehensive environmental flow assessments, but are largely restricted in focus to the flow-related habitat requirements of specific in-stream target species, for various biological activities and times of the year. Outputs in the form of flow-habitat time series and exceedance plots enable the environmental implications of alternative hydrological scenarios to be explored, and can incorporate many aspects of flow variability. Moreover, as most of these methods are designed to focus on flow-related changes in the suitability of physical habitat for freshwater fish species, they are well suited to detailed modeling of the requirements of indicator fish species or reproductive guilds. However, the models tend to represent flow-habitat relationships in a relatively simplistic way (in relation to few hydraulic variables), they do not predict biotic responses (e.g., population biomass, or increase/decline) to flow regulation, and do not take into consideration the requirements of other components of the ecosystem and their interactions, including those with fish (Pusey 1998). Consequently, they are considered more appropriate as one of a wider set of tools for addressing the flow requirements of entire river systems and their fisheries.

TABLE 3.
Flow categories that are reduced, or increased, in magnitude or number, to produce described consequences, and the five ecosystem components for which consequences are routinely predicted in DRIFT.

Flow categories	Consequences described for:	Ecosystem component
1. Dry-season low flow (range of low flows)	4 levels of increase	1. Fluvial geomorphology
2. Wet-season low flow (range of low flows)	or decrease	2. Water quality
3. Intra-annual floods: Class 1	4 changes in the	3. Plants
4. Intra-annual floods: Class 2	number per annum	4. Aquatic invertebrates
5. Intra-annual floods: Class 3		5. Fish
6. Intra-annual floods: Class 4		The hydraulics of the
7. 1:2 year flood	Presence	river channel are
8. 1:5 year flood	or absence	also computed.
9. 1:10 year flood		
10. 1:20 year flood		

Source: King et al. 2003

Although a relatively small proportion of the global pool of methods, holistic methodologies by their very nature as whole-ecosystem approaches provide the most appropriate means of assessing environmental flows for fisheries in complex river systems. Earlier techniques of this type, including the expert panel assessment methods (e.g., EPAM and SPAM), tend to provide outputs of lower resolution and more limited scope in terms of the ecosystem attributes considered and flexibility, than the most well-developed group comprising the Flow Restoration Methodology, BBM, DRIFT and the Benchmarking Methodology. This latter group essentially represents frameworks for the structured collation and interpretation of complex, multi-disciplinary information drawn from a large range of sources, including hydrological, environmental, and in a few instances (notably in the BBM, DRIFT and Adapted BBM-DRIFT) also social data pertaining to the dependencies of local communities on natural riverine resources (including but not exclusive to fisheries).

The outputs generated from such tools, in the form of hydrological scenarios with attendant geomorphological, ecological, and social implications, are sufficiently predictive and flexible that they can be used alongside other water and land resource development scenarios, to examine and evaluate tradeoffs in basin water allocation between other water use sectors, notably agriculture, biodiversity conservation and fisheries. Such approaches are therefore especially valuable in floodplain rivers where the seasonally inundated floodplain is used for a number of livelihood strategies in addition to fisheries.

It is noteworthy that although the Managed Flood Release Approach of Acreman et al. (2000) (see section: *Review of Environmental Flow Methodologies),* based on experimental flood releases from dams, has been used in river floodplain restoration projects in Africa to sustain fisheries production and dependent livelihoods, it is only partially holistic in nature. It does not appear to deal with the requirements of the ecosystem for a range of hydrological conditions other than large flood events, and does not provide an explicit set of methods and tools for calculating the ecological flow needs of the river system of concern. EPAM, also based on observation of the ecological consequences of experimental flow releases from dams (Swales and Harris 1995), is likewise insufficiently comprehensive to address the needs of floodplain river ecosystems.

As is the case with habitat simulation methods, most applications of holistic approaches have been undertaken primarily in upland, temperate rivers with limited floodplain development, and in relatively small basins. Of the holistic methodologies available, only the Benchmarking Methodology has addressed the environmental water requirements of large, tropical floodplain systems, including the flows required to sustain the floodplain, estuarine ecosystem and nearshore fisheries (e.g., the Burdekin River, North Queensland, Australia). This methodology has thus far only been used to define the limits to new water resource development, but has excellent potential to define opportunities for flow restoration in exploited rivers.

The Flow Restoration Methodology in turn is specifically designed to provide detailed assessments of the impacts of past and extant flow regulation, to identify opportunities for flow restoration and to quantify the hydrological components that should be restored to achieve particular geomorphological and ecological objectives. The Flow Restoration Methodology can be applied within the framework of a broader benchmarking exercise, such that together these methods provide guidance on limits to flow modification as well as targets for flow and ecological restoration in particular parts of a regulated catchment. Benchmarking cannot be used in catchments with very little water resource development as it is these developments that provide the benchmarks for assessing the risk of adverse ecological changes following flow regulation in that or a similar catchment. Adopting

benchmarks from other catchments, even very similar ones, is a possibility but is considered a risky procedure until the commonalities of ecological responses to flow regulation are better understood (see Arthington et al. 2006, for further comment).

Many of the technical tools within the Flow Restoration Methodology have been incorporated into DRIFT (e.g., most of the field and analytical methods involved in the fish component of DRIFT). Additional features of the Flow Restoration Methodology are the capacity to include consideration of non-flow related impacts on the aquatic ecosystem, and to give advice on how the river ecosystem as a whole could be restored to some degree by various measures, of which flow restoration is only one option and not always one that can be achieved to the full in a regulated river system for technical, economic, social and legal reasons. Such considerations could also be built into the DRIFT framework, as discussed also in section *River Fisheries Models* in relation to the range of issues now being considered in the assessment of impacts on floodplain fisheries.

Benchmarking and DRIFT appear to be the only holistic methodologies incorporating methods to predict the implications (or assess the risk) of flow regulation for river/nearshore fisheries, and DRIFT takes this further by assessing the implications for all types of use of river resources by people (e.g., food and fibre derived from aquatic and riparian vegetation). Unlike DRIFT, Australian holistic methodologies do not involve assessment of the social and economic implications of water resource development and flow regulation, but they do provide explicit inputs to related decision–making processes involving appraisal of all of the environmental, social and economic costs and benefits of water use. In other words, DRIFT has existing capacity to develop evaluations of ecosystem goods and services and to predict how water resource development, or flow regime restoration, may impact on goods and services and the people dependent upon them.

Few of the methodologies described above have been subjected to rigorous validation procedures or tests of the reproducibility of their outputs, although efforts to do so are in progress for the BBM, Benchmarking Methodology and DRIFT. Studies incorporating comparisons of the outputs from two methodologies are very rare. Two are in progress in Australia (BBM *versus* Benchmarking, Flow Restoration versus Benchmarking) and one has been conducted in South Africa (BBM *versus* DRIFT).

Examination of the strengths and deficiencies of the most comprehensive group of holistic methodologies available (Table 2) suggests that DRIFT is the methodology that is currently most amenable to further modification for consideration of flow requirements for complex river fisheries. However, to apply DRIFT in large floodplain river systems where the livelihoods of local communities depend directly and principally upon rivers for water resources and major floodplain fisheries composed of many species, will require further development, refinement and field trials to accommodate the added spatial complexity and higher biological diversity of these ecosystems. In addition, for DRIFT to be able to reflect the particular importance of river fisheries in these larger river systems, it will need to incorporate recent developments in fisheries production modeling. Finally, much work remains to be done on the phases of environmental flow assessment dealing with scenario development and tradeoff analysis, to enhance the role of holistic methodologies in the allocation of water in river basins. These issues of tool adaptation and integration are discussed further in the following sections.

The DRIFT Methodology Further Examined

The DRIFT Methodology

Introduction

DRIFT (Downstream Response to Imposed Flow Transformation) is an interactive, "holistic" (Arthington et al. 1992; Tharme 1996) approach to advising on environmental flows for rivers. It was developed from earlier holistic methodologies such as the Building Block Methodology (BBM)(King and Louw 1998), through several applications in southern Africa (e.g., King et al. 1999; Brown et al. 2000a, 2000b, 2006; Brown and King 2002). It is described in detail in King et al. (2000b, 2003), Arthington et al. (2003), Brown and Joubert (2003.) and Brown et al. (2005, 2006).

The central rationale of DRIFT is that different categories of flow (wet and dry season low flows; small, medium and large floods) maintain different parts of the river ecosystem. Thus, manipulation of one or more of these categories of flow will affect the ecosystem differently than manipulation of some other combination (King et al. 2000b, 2003). Furthermore:

- these categories of flow can be identified and isolated from the historical hydrological record,
- the probable biophysical consequences of manipulation of a single flow category can be described,
- once these consequences have been described for all flow categories, flows can be re-combined in various ways and the overall impact on river condition of each new flow regime derived,
- the change in river condition can be indicated as a change in river management class,
- the implications of the change in river condition for common-property users of the river's resources can be described.

The main features of the DRIFT process are:

- taking the present-day flow regime as a starting point, ecosystem changes linked to a range of flow manipulations are predicted,
- a database of these predictions is compiled, which is queried to produce biophysical scenarios of the ecosystem changes linked to any contemplated flow manipulation,
- the biophysical scenarios are taken further to describe social impacts for common-property subsistence users of the river (the Population at Risk or PAR).

Outline of DRIFT

DRIFT Modules

DRIFT consists of four modules (biophysical, subsistence use, biophysical and social scenario development and social compensation economics) (King et al. 2000b). In the first, or biophysical module, the river ecosystem is described and predictive capacity developed on how it would change with flow changes. In the second, or subsistence module, links are described between riparian people who are common-property subsistence users of river resources, the resources they use, and their health. The objective is to develop predictive capacity of how river changes would impact their lives. In the third module, scenarios are built of potential future flows and of the predicted impacts of these on the river and the riparian people. The fourth, or compensation-economics, module lists compensation and mitigation costs (King et al. 2000b). Although all four modules have been applied (e.g., King et al. 1999; Sabet et al. 2002), the first and third modules can be applied alone (e.g., King et al. 1999; Brown et al. 2000a, 2000b) and are the most developed.

Imparting structure to the DRIFT process

DRIFT has several characteristics that impart structure to specialist deliberations on the consequences of flow changes (King et al. 2000b):

Environmental Flow Sites

Data collection and subsequent deliberations are centered on river sites, each of which is representative of a river reach.

Daily hydrology

The present-day long-term daily flow data for each site are separated into ten flow categories and specialists predict the consequences of up to four levels of change from present condition in each flow category for different components of the river ecosystem.

Multi-disciplinary team

The specialists routinely included are in the following disciplines: sedimentology/fluvial geomorphology, water quality, plants, aquatic invertebrates and fish. Depending on the river under study additional components, such as mammals, birds and herpetofauna, can be added. The specialists build up a picture of predicted change to any presented flow manipulation, starting with channel changes, then water quality and temperature, then vegetation, invertebrates and fish. Other disciplines included where relevant are inserted into this sequence as appropriate.

G*eneric lists*

When recording the consequences of each considered flow change, the specialists consider any number of sub-components that may be relevant to their ecosystem components. These are contained in Generic Lists for each component.

For each considered flow change at each study site, the effect on each selected item on the Generic List is described. Each specialist's list may consist of any number of items, but usually four to less than twenty (Table 5). For each flow reduction at each study site, the effect

TABLE 4.
Examples of entries in generic lists for an environmental flow study using DRIFT, based on the Lesotho Highlands Water Project.

Discipline	Generic list entry	Description of the links to flow (upper entry) and social relevance (lower entry)
Geomorphology	Deposition of colloidal material	Minimum velocity for maintenance of movement of colloidal material in main channel = 0.05 m s^{-1}. Muddy areas are linked to loss of cobble habitat, increased algal growth, bogging of livestock, and gastric illnesses.
Water quality	Nutrient levels	Nutrient levels in pools increase under low flow conditions. Water in pools flushed by > Class II floods. High nutrients encourage algal growth, which is linked to increased incidence of diarrhea in people and livestock, and loss of cobble habitat.
Vegetation	*Chenopodium album*	Found mostly in the wetbank vegetation zone, the width of which is reduced by a reduction in the volume and variability of low flows and in the number of Class I floods. Abundance is affected by narrowing the zone. Important source of firewood. Also used as a medicine.
Fish	Maluti Minnow	Inhabits quiet, shallow waters in rocky reaches with high water quality IUCN Red Data Book rare species. Restricted to the Highlands of Lesotho. Threatened with extinction.
Invertebrates	*Simulium nigritarse*	Filter feeder in slow, eutrophic water. Blood-sucking pest of poultry.

Source: King et al. 2002b

TABLE 5.
Severity Ratings for each prediction of flow-related change in DRIFT.

Severity rating	Severity of change	*Equivalent loss* (abundance/concentration)	Equivalent gain (abundance/concentration)
0	None	no change	No change
1	Negligible	80-100% retained	1-25% gain
2	Low	60-79% retained	26-67% gain
3	Moderate	40-59% retained	68-250% gain
4	Severe	20-39% retained	251-500% gain
5	Critically severe	0-19% retained; includes local extinction	501% gain to ∞: up to pest proportions

Source: King et al. 2003

on each item on each list is described. The list items may include anything that increases or decreases in abundance or size, such as species or habitats, and are chosen based on their known susceptibility to flow changes, their role as key species or features, or their relevance to subsistence users. In the Lesotho Study (King et al. 2000b), about 130 items were included in the various lists and, with all eight study sites and flow-change levels considered, more than 20,000 consequences were recorded. These were entered into a custom-built database.

Severity ratings

Each consequence is accompanied by a Severity Rating (Table 5), which indicates (1) if the sub-component is expected to *increase* or *decrease* in abundance, magnitude or size; and (2) the *severity* of that increase/decrease, on a scale of 0 (no measurable change) to 5 (very large change). The scale accommodates some uncertainty, as each rating encompasses a range in percentage gain or loss. Greater uncertainty can be expressed through providing a range of severity ratings (i.e., a range of ranges) for any one predicted change (after King et al. 2000b).

Integrity ratings

To assist with the eventual placement of flow scenarios within a classification of overall river condition, the Severity Ratings are taken further to indicate whether the change would be a shift toward or away from the natural condition. The Severity Ratings hold their original numerical value of between 0 and 5, but are given an additional negative or positive sign, to transform them from *Severity Ratings* (of changes in abundance or extent) to *Integrity Ratings* (of shift to/away from naturalness), where:

- *toward natural* is represented by a positive Integrity Rating; and
- *away from natural* is represented by a negative Integrity Rating.

The DRIFT Biophysical Database

Data contained in the database

The output of the DRIFT specialist sessions is a matrix of consequences, completed by the specialists, for a range of possible reductions (or additions) in the ten flow categories. These data are entered into the DRIFT database (Figure 1), together with information on the data sources used.

In summary, each entry within the database consists of (Table 6):

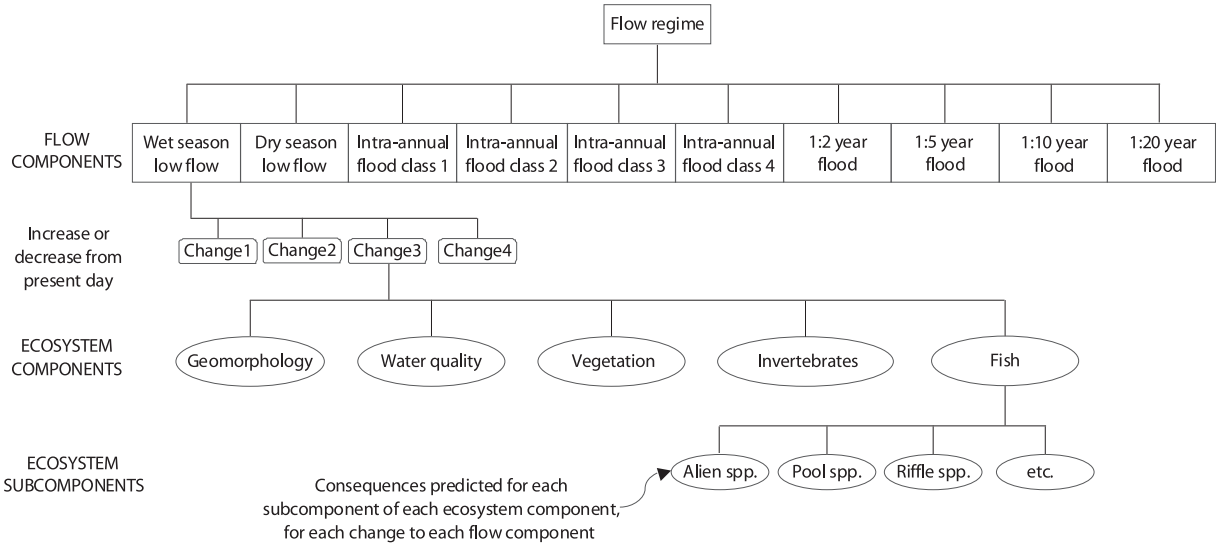
- A site name
- A flow reduction from (or addition to) the present-day status of one of the low or high flow categories (e.g., presently an average of four Class 2 floods per annum: reduce to two per annum);

TABLE 6.
Example of a consequence entry in the DRIFT database for one ecosystem sub-component.

Type of information	Information
Site	2
Flow reduction level	Reduction level 4 of dry-season low flows
Component	Invertebrates
Sub-component	Simulium nigritarse
Direction of change in abundance	Increase
Severity rating	5: critically severe
Integrity rating	–5: away from natural
Ecological significance	Filter feeder in slow, eutrophic water
Social significance	Blood-sucking pest of poultry
Volume of water	12 m^3 x 10^6 per annum

- The consequences of this for a range of ecosystem components (e.g., plants) and their sub-components (e.g., algae), expressed as:
  - The direction of predicted change (increase or decrease in abundance);
  - The extent of change (Severity Rating);
  - The expected impact on river condition, relative to natural (Integrity Rating);
- Descriptions of the ecological and social significance of the predicted change;
- The volume of water required to deliver this flow, expressed as m^3 x 10^6 per annum for each of the ten flow classes, per season and per annum.

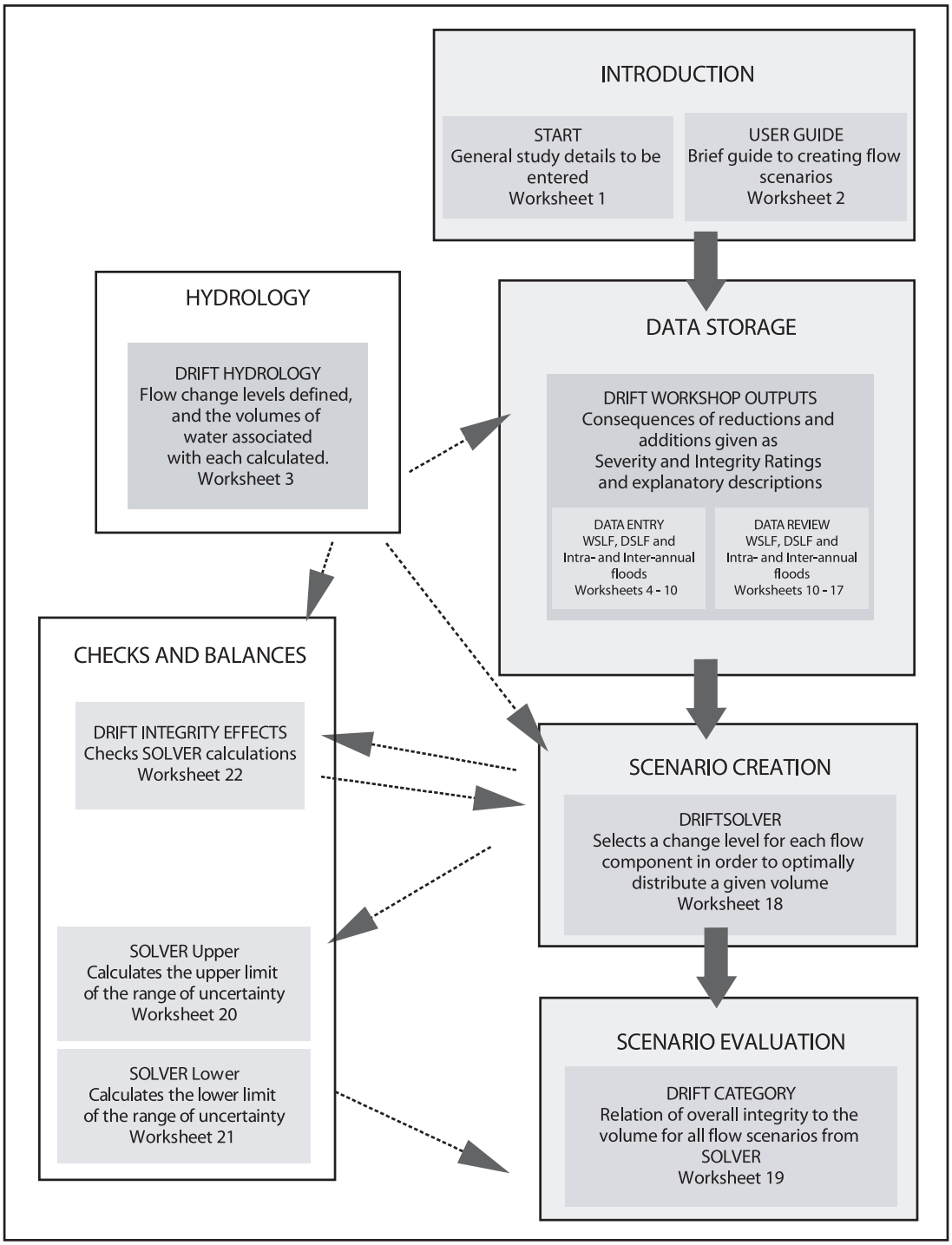
FIGURE 1.
Framework for consequences of reductions (or additions) in low or high flows for ecosystem sub-components of DRIFT.



Structure of the database

The DRIFT database comprises six Excel worksheets (Figure 2) that can loosely be divided into two groups: data storage, and scenario creation and evaluation (Brown and Joubert 2003). In the second group, integer linear programming (Winston 1994) is used to re-combine a selected change level for each flow category into a modified flow regime and describe its consequences, using the software DRIFTSOLVER and DRIFT CATEGORY.

FIGURE 2.
The Excel worksheets within the DRIFT database.



Source: from Brown et al. 2005

In DRIFTSOLVER, the integer linear program optimizes the distribution of a given total volume of water among different change levels of flow categories in a way that results in the lowest aggregate impact on the riverine ecosystem according to the Integrity Ratings. It does this by summing the Integrity Ratings of all the sub-components, taking into account all the negative or positive signs, to produce combinations of high and low flows that return the highest possible overall Integrity Score for that volume.

DRIFT CATEGORY provides a summary of many scenarios, showing the relationship between overall river condition and volume of water remaining in the river. The shape of the plot is specific for the river site under its present flow and management conditions, and is based on the 'least-damaging' mix of high and low flows. The plot can be used to examine the relationship between volume of water and ecosystem integrity, and to identify features such as inflection points where a small change in flow is linked to a large change in integrity status. Although DRIFTSOLVER automatically searches for the optimal distribution of low and high flows for any volume of water, DRIFT CATEGORY can be used to indicate the implications for river condition of non-optimal distributions of flows, such as may happen where large floods (e.g., greater than Class 2 floods) cannot be released through an upstream dam.

The DRIFT CATEGORY plots are intended for use in decision-making. The level of detail they provide is sufficient to inform the broad-level tradeoffs considered by decision makers when balancing potentially conflicting uses such as environmental protection *versus* agricultural development. The summary plots are, however, backed by the detailed consequence data provided by the specialists.

Detailed biophysical descriptions

Compilation of the detailed description of change for any flow scenario is not yet automated. When a flow scenario, with its descriptions, is compiled (whether manually or automated), this requires interpretation by one or more experienced river scientists as a quality-control measure. Examples from a detailed description are given in Table 5.

Links to the subsistence and economic modules

The predicted changes in river condition could affect the lives and livelihoods of common-property subsistence users of the river (the People at Risk - PAR). Thus, the biophysical scenarios created by DRIFTSOLVER are used as the template for developing social scenarios and their economic implications. For each scenario, site and resource, the impacts are ranked as None, Low, Moderate and Severe, using expert opinion and considering the following issues (Sabet et al. 2002).

- The level of predicted change of a resource
- Importance of the resource for the livelihoods of the affected populations
- Number of households harvesting the resource
- Frequency of usage of the resource
- Availability of alternative resources

The categories of social impact are broadly defined as follows:

None:	No appreciable change expected.
Low:	The resource is not important or, if important, its quantity is predicted to change by less than 20 percent.
Moderate:	The resource is important, and its quantity is predicted to change by 20-50 percent.
Severe:	The resource is considered essential for the livelihoods of the PAR; and it is used by more than 20 percent of the PAR households; and the predicted biophysical change is greater than 50 percent.

The predicted impacts on human health of each of the scenarios takes into consideration:

- the wide range of factors influencing health in the PAR, some of which will have no links to the river;
- the extent of river use by members of the PAR; and
- any predicted biophysical changes that could influence people's health.

Table 7 provides an example of the considerations that were used to create the link between biophysical changes in the river and a listed PAR health concern. The column 'Ecological link' lists some parts of the ecosystem that could change with a flow change, and an explanation of how this could affect people and their livestock. The weighting indicates the importance of this to the health issue under discussion. The 'Onset' entry indicates the time span over which the impacts described in column 1 may become apparent. The likelihood of the ecological conditions described in column 2 developing is given in the biophysical scenarios developed in DRIFTSOLVER.

The fish component of DRIFT

DRIFT offers a structured process for predicting the biophysical, social and economic consequences of altering a river's flow regime. The fish component of DRIFT is a fully compatible, 10-step protocol designed to make such predictions using field data on a river's fish fauna linked to information on flow-related aspects of fish biology drawn from the literature and the knowledge base and professional experience of fish ecologists.

TABLE 7.
Examples of health-related ways a river ecosystem could change, and the relevance of this for diarrhea and eye and skin diseases among the People at Risk (PAR) in the DRIFT methodology. Columns 1 and 2 should be considered together.

Ecological link	Weighting	Onset (years)
Colloids An increase in colloidal material allows diarrheal, disease-causing organisms such as Giardia to remain in the river for longer, thus increasing the chances of people becoming infected either through contact (skin and eye infections) or consumption (diarrheal disease).	High	2-10
Total dissolved solids Drinking turbid water with high TSS levels does not necessarily have direct health effects, but such effects can occur when infectious disease agents adsorb onto the particulate matter.	High	2-10
Algal blooms High summer flows flush algae from the river, but low winter flows allow the plants to accumulate in quiet areas. Loss of flushing flows will increase the risk of algal blooms, with resulting adverse health effects through swallowing algae-contaminated water.	High	1-2
Blackflies Increases in the numbers of blackflies can result in an increase in the level of irritation caused by their bites. Blackflies can also carry disease from faeces to food or utensils used by humans (faecal – oral route).	Low	1-2

Source: Sabet et al. 2002

Note: Columns 1 and 2 should be considered together

The basic steps in the fish component of DRIFT are the following:

Step 1: Review of literature to produce a compilation of published flow-related and other information about each fish species recorded in the river under investigation.

Step 2: Selection of study sites (in collaboration with the full team of DRIFT specialists) to characterize river reaches likely to be affected by existing and future water resource developments.

Step 3: Seasonal field surveys at each site to determine fish species composition, abundance, habitat use, life history, diet, etc., in relation to existing flow conditions.

Step 4: Analysis of field data to generate habitat preference curves for each fish species.

Step 5: Tabulation of field data and information from literature review to produce a summary of flow-related data about each fish species.

Step 6: Development of scenarios of flow regime change for evaluation using DRIFT.

Step 7: Development of protocols to document the consequences of flow regime change for each fish species at each study site.

Step 8: Prediction of the ecological and social consequences of flow regime change for each fish species at each study site.

Step 9: Preparation of a monitoring strategy to assess the outcomes of environmental flow provisions.

Step 10: Implementation of monitoring program, evaluation of ecological outcomes of any environmental flow provisions, and adjustment of those provisions in the light of new knowledge generated by monitoring (and research).

Steps 1-8 are described in Metsi Consultants (2000a, 2000b) and in a worked example (Arthington et al. 2003). Steps 9 and 10, the monitoring strategy and its implementation, are also described in Metsi Consultants (2000b).

The fish component of DRIFT is presently designed to predict the consequences of four types of flow reduction: loss (or gain) of low flows during normally dry months; loss (or gain) of low flows during normally wet months; loss (or gain) of intra-annual floods; and presence or absence of inter-annual floods with return periods of 1:2, 1:5, 1:10 and 1:20 years (see Table 3 above). Each change in a flow characteristic is evaluated separately in terms of the likely ecological consequences for each fish species, and the attendant social consequences are also evaluated.

Two generic lists present the flow related issues affecting fish. These are: (1) the ecological requirements of fish likely to be affected by reduction in low flows, and (2) the ecological requirements of fish likely to be affected by reduction (or loss) of intra- and inter-annual floods. The possible effects of flow regime change on fish may be expressed for individuals, populations or species interactions (Table 8). The ultimate ecological consequence of flow regime change was considered to be the loss of one or more species of fish from a study site and the river zone it represented. All generic issues were identified from the literature and researcher's experience of river ecology and environmental flow studies conducted in Australia and South Africa.

Ecological consequence summaries for fish are entered in the DRIFT database as shown in Table 4 above, and the social implications of these ecological changes are also rated for each fish species present at a site and likewise, are entered into the DRIFT database.

The level of confidence in each fish assessments made using DRIFT is rated according to the information sources available and their scientific quality, thus providing the water manager/client with an explicit means to undertake his/her own assessment of the risks

TABLE 8.
Effects of flow regime change on fish individuals, populations and species interactions (from the fish component of the DRIFT methodology).

Effect code	Effect on fish
A	Change in fish health (due to effects of disease, parasites, toxic algae, etc.)
B	Change in fish body condition (size, weight)
C	Change in abundances of adult fish
D	Change in abundance of juvenile fish
E	Change in abundance of larval fish
F	Change in spawning and hatching success
G	Change in level of passage and dispersal for: (i) Adults (ii) Juveniles (iii) Larvae
H	Change in predation level
I	Change in competitive pressure
J	Change in mortality levels
K	Loss of species from site

Source: Arthington et al. 2003

associated with management actions based on limited or low quality information. The rating scheme applied in the fish component of DRIFT closely resembles that recommended by Cottingham et al. (2002) and Downes et al. (2002) as "levels of evidence that support environmental flow assessments."

The developers of the fish component of DRIFT (Arthington et al. 2003) recommend that the predictions related to the various biophysical components could be strengthened by embedding qualitative and quantitative methods and models linking the biological requirements of fish to the hydrological and biophysical characteristics of rivers. For example, the generic lists presenting the ecological requirements of fish likely to be affected by reduction in low flows, and those likely to be affected by reduction (or loss) of intra- and inter-annual floods, could be represented as qualitative models using various diagrammatic formats. Since these models represent the current state of knowledge of such relationships, drawn from the literature and personal experience, they represent both a qualitative knowledge summary (explicit references to published literature, or to personal experience must be suitably annotated) and an education/communication tool for users of DRIFT. Indeed, qualitative models should provide the foundations for the development of Bayesian models, since the essence of Bayesian networks consists in defining the system studied as a network of variables linked by probabilistic interactions (Jensen 1996). In large floodplain rivers, data exist to facilitate quantitative modeling of floodplain-fish relationships (e.g., Arthington et al. 2005; Balcombe et al. 2006) and studies discussed below.

These are other recommendations for further innovation and research to further develop DRIFT which are outlined in Arthington et al. (2003) and below.

Verification of results achieved using DRIFT

To date, DRIFT has been used in a predictive capacity, and insufficient time has passed for the accuracy of its predictions to be verified through monitoring. However, monitoring programmes aimed at verification of DRIFT biophysical and social predictions have been approved for the rivers downstream of Lesotho Highlands Water Project Phase 1 structures (LHDA 2003), and a programme for the verification of DRIFT biophysical results has been proposed for the Palmiet River, in the Western Cape, South Africa (Brown et al. 2000a, 2000b).

Additional support for the rationality of the results obtained using DRIFT was also forthcoming from a dual application of the BBM and DRIFT methodologies at three sites on the Breede River, Western Cape, South Africa, where the same team of scientists used the same datasets to provide environmental flow scenarios (Brown and King 2002). The results obtained for the two methods were very similar in terms of: the percentage Mean Annual Runoff (MAR) required to facilitate a desired river condition and the temporal distribution of the annual volume of water allocated to the environmental flow.

A detailed analysis of the dual application, and the relative similarities, difference, advantages and disadvantages of the BBM and DRIFT that it illustrated is currently being compiled (King et al. 2003). Other recent developments of DRIFT are presented in Brown and Joubert (2003.) and Brown et al. (2005, 2006).

Research needs

The following aspects and concerns have been raised as possible areas for future development and research to refine DRIFT.

User manual

Write a manual for DRIFT (funds granted by the South African Water Research Commission, for 2003/2004).

Generic lists

Expand and refine the drop-down lists of components, sub-components, elements and sub-elements. These lists exist in draft form for upland rivers, funded by the South African Water Research Commission. Generic lists need formulating for lowland rivers, floodplains and estuaries.

Hydrology

Explore additional hydrological parameters and statistics that express variability. Develop new hydrological parameters for generic lists.

Large rivers – longitudinal and lateral interactions

Develop a process for the inclusion and consideration of longitudinal biological and hydrological interactions. Issues to include: the need of a species to migrate, the extent and route of that migration, which life stages migrate, large-scale processes such as invertebrate drift, and lateral channel-floodplain exchanges of materials, energy and biota, and provision of fish passes to facilitate migration of fish affected by hydraulic structures.

Consideration of water quality

Most discussion of environmental flows is currently confined to quantity of water. However, increased attention needs to be given to including consideration of water quality. DRIFT should address this.

Interactions between biophysical components

Develop a process to assist biologists in interpreting information from hydrologists, geomorphologists and aquatic chemists about the consequences of flow change, to determine the implications of these changes for their disciplines.

Variability

Build in natural variability and natural changes in abundance of biota. This may be possible using ‘gates’ in the system (e.g., and/or and if/then logical statements). Explore the possibility of using a Bayesian approach.

Weightings

Assign weights that reflect the contributions of different sub-components to overall river condition.

Validation

Calibrate DRIFT CATEGORY using monitoring results. There is also a need to check how additive and/or synergistic effects of flow modifications would be dealt with in the model.

Automation

Explore semi-automation of the detailed results, possibly through a Bayesian approach. Develop and semi-automate the social component. Incorporate subsistence use data into DRIFTSOLVER.

Floodplain rivers and fisheries

Develop new hydrological parameters and determine appropriate time steps for floodplain rivers. For instance, there is some indication that it may not be necessary to quantify the spatial array of habitats in large rivers: multi-species approaches may be sufficient for environmental flow concerns related to fisheries.

Anthropogenic effects

Explore possible inclusion of feedback loops that will affect ecosystem functioning, such as:

- effects of exploitation of riverine resources;
- losses that affect social groups differently; and
- effects of land-use on hydrological patterns.

Other models

Explore the scope for inclusion of conceptual models depicting/representing biophysical responses to natural flow components and to flow modifications (e.g., benchmarking models from Australian studies in large floodplain rivers). These models could be used to illustrate the issues presented in generic lists, thus aiding users of DRIFT in their summation and application of local knowledge.

Models such as PHABSIM (Bovee 1982; Stalnaker et al. 1994) could be applied to predict fish responses to flow modification in some river channels. Univariate and multivariate statistical models of the environmental factors and flow attributes influencing fish populations and assemblage structure are also proving useful in the design of environmental flow regimes for fish in Australian rivers (e.g., Kennard 2000; Kennard et al. 2006a, 2006b; Pusey et al. 2000; Arthington et al. 2005; Balcombe et al. 2006). Other types of modeling approaches are discussed in section: *River Fisheries Models* below.

Incorporating Bayesian Networks into DRIFT

Bayesian networks have recently become quite popular among holistic models dealing with ecosystem issues. The principles and current applications of these networks of variables probabilistically interconnected are detailed in section: *Bayesian Networks.* The need for DRIFT to handle a large amount of information of ecological nature, involving multiple variables, interactions and feedback loops between these variables (whether species, hydraulic or physicochemical), has called for an integrative tool able to deal with networks. The paucity of quantitative data about these interactions and their probabilistic nature has led to the idea of integrating Bayesian networks into DRIFT. We detail below the differences between these methods, as well as the constraints and added value of their integration.

Assessment of the merits of the two approaches

DRIFT and Bayesian networks differ both in the way they manage data, and in the information they convey. Bayesian networks are conceptual in nature, and do not deal with limited variations, i.e., a 100 percent chance of 20 percent change (DRIFT) is different from a 20 percent chance of change (Bayesian). Thus, the manner in which data are recorded and analyzed for flow regime optimization in DRIFT offers some advantages over that in Bayesian networks. The extra information in DRIFT is that the direction and extent of possible change are provided, along with confidence limits, whereas Bayesian theory provides a direction and a probability of an unknown level of change.

The relative strengths and weaknesses of the two approaches are:

1. The hydrological manipulations and optimization in DRIFTSOLVER are not easily handled using the Bayesian approach, and it seems neither practical nor advantageous to replace the process used in DRIFTSOLVER with a Bayesian approach.

2. One of the key activities in DRIFT is population of the database with specialists' inputs on flow-consequence relationships. The more structured and standardized this activity, the more rigorous and repeatable will be the process. Several approaches to structure this have been implemented (e.g., King et al. 2003) but this does seem an area where Bayesian networks could be useful. Bayesian networks could guide the deliberations of specialists by forcing them to deal with all possible outcomes of interactions between variables. Note that the conceptual models used in the Benchmarking Methodology are essentially the same in concept to Bayesian networks (but without the assignment of probabilities) and should inform network development (see section: *Verification of Results Achieved using DRIFT).* The Bayesian networks software is also highly visual and dynamic, making complex information easier to interpret. The relevant variables must still be identified and the value levels determined, and the DRIFT generic lists of sub-components and conceptual models would be critical in ensuring all relevant variables are identified.

3. Bayesian networks become complex quickly as new variables are added, and parameterization of the probability tables (by the specialists) can become extremely complex. Thus, there is a need to explore ways of incorporating Bayesian networks as small modular units (each representing a flow-fish conceptual model, for example), which could be parameterized individually.

4. DRIFT severity and integrity scores have been applied in several studies and work well. Furthermore, they are integral to the DRIFTSOLVER optimization process. Thus, to avoid requiring specialists to score impacts in two different ways, it is desirable to retain basic DRIFT type scoring for the driving variables of the system. There is therefore a need to explore ways of ensuring that the Bayesian networks developed are compatible with DRIFT scoring.

5. In Bayesian networks, the weighting is implicit. The person who parameterizes each of the probability tables may weigh different components in their mind but the weights are not recorded, nor are there any guarantees that they are applied consistently. In DRIFT, explicit weighting systems are used and recorded at all levels. Thus, if the two methods are to be integrated, procedures for explicit weighting and recording will need to be incorporated into the Bayesian networks in order to retain the consistency and transparency of the DRIFT process.

Proposed procedure to aid prediction of the consequences of flow change

The possibility exists to use Bayesian networks to assist with determination of the consequences of flow changes for ecosystem sub-components in DRIFT. As a first step, separate networks could be developed to assist biologists in interpreting information from geomorphologists and aquatic chemists about the consequences of flow change, to determine the implications of these changes for their disciplines.

In line with a modular approach, a series of conceptual models and small Bayesian networks for each flow class, and each ecosystem sub-component of concern, could be constructed. The inputs to the Bayesian module could be scores of DRIFT consequences (i.e., severity and direction of change) as this would (1) ensure compatibility with the DRIFT database, and (2) simplify the linkages between modules. The output of the Bayesian module would be probabilities of a combination of consequences of flow change resulting in optimal or sub-optimal conditions for the sub-component of concern. These probabilities could then be used to guide the predictions of the consequences of flow change for the sub-component.

For instance, changes in wet season low flows (WSLF) may have implications for a minnow population in a river via its influences on a number of variables affecting different life-stages of the minnow (see Arthington et al. 2003). These variables could include: salinity, temperature, oxygen levels, riffle area, depth and velocity, and supply of food. Using Bayesian networks, and the DRIFT scoring system (increase or decrease of severity 1-5) it is possible for the minnow specialist to see how the consequences of change in a range of variables could affect the minnow. The relevant boxes for salinity and riffle area are shown in Level 1 of Figure 3.

For each individual variable, e.g., salinity and riffle area in Figure 3, a variable-specific node could be generated for each ecosystem sub-component of interest that considers whether the predicted changes would result in conditions that were optimal, sub-optimal or lethal for that sub-component, e.g., minnows in Figure 3 (Level 2). These nodes are comprised of probability tables (Table 9) that should be parameterized by the specialist using information that they have gathered on the tolerance of minnow to the

FIGURE 3.
Hypothetical Bayesian network constructed for DRIFT consequences of changes in wet season low flows for minnows, using two 'driving' variables, salinity and riffle area.

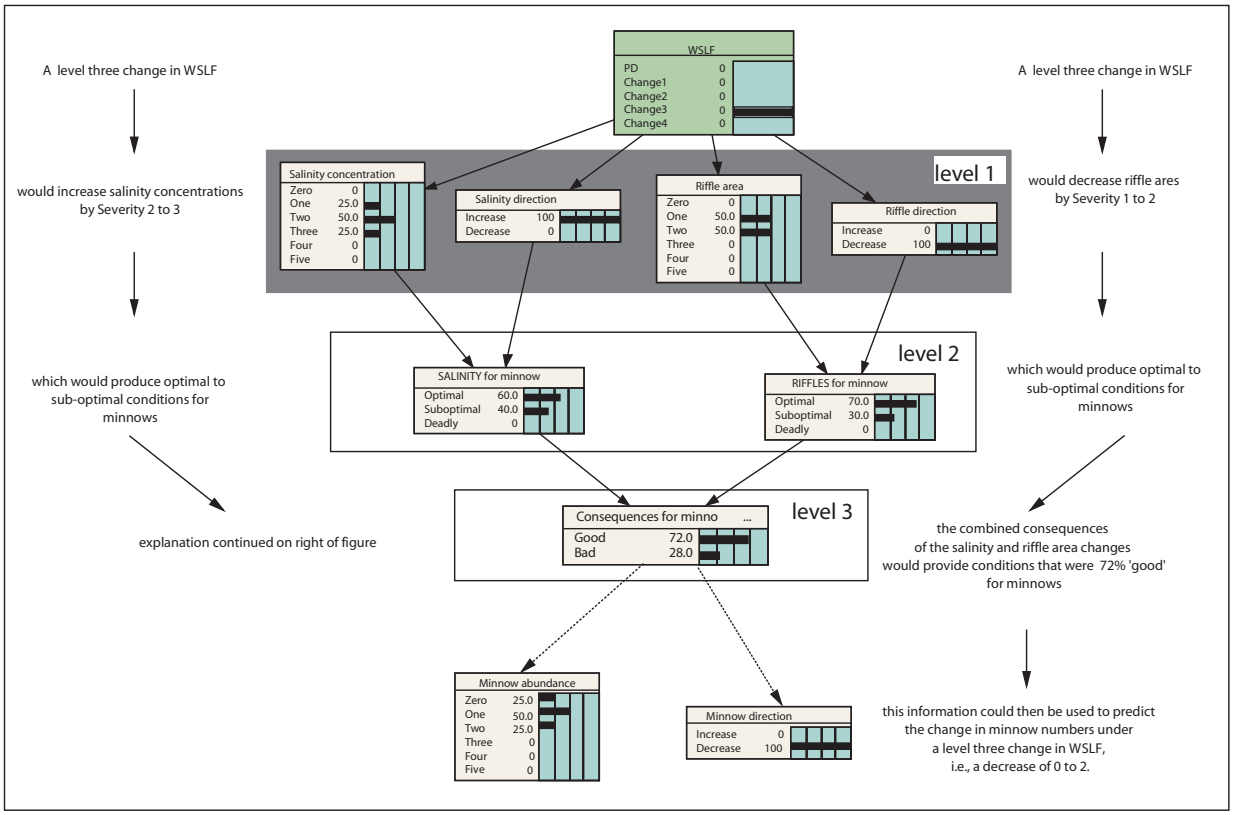


TABLE 9.
An example of the possible probabilities in the variable node (Level 2 in Figure 3), based on a hypothetical Bayesian network constructed to show the consequences of changes in wet season low flows for minnows, using two 'driving' variables, salinity and riffle area.

Magnitude	Direction of change	For sub-component (e.g., minnow)		
		Optimal	Sub-optimal	Lethal
Zero	Increase	100	0	0
One	Increase	100	0	0
Two	Increase	100	0	0
Three	Increase	0	100	0
Four	Increase	0	0	100
Five	Increase	0	0	100
Zero	Decrease	100	0	0
One	Decrease	100	0	0
Two	Decrease	100	0	0
Three	Decrease	0	100	0
Four	Decrease	0	100	0
Five	Decrease	0	0	100

variable and must take account of both the direction (increase or decrease) of change and the magnitude of that change (severity 1-5). Variable-specific nodes can be added for all the flow-related variables likely to dictate the response of the minnow.

To assess the overall consequences for the minnow of the changes in all the variables considered another node would need to be generated (Level 3, Figure 3), which provides an indication of whether the combination of predicted changes is likely to result in conditions that are beneficial or detrimental to the minnow. This information can then be used to assist the specialists in deciding on the severity and integrity ratings for DRIFT.

Ways by which the allocation of probabilities at Level 3 can be automated or semi-automated will need to be explored because there will be many combinations to consider, making allocation of probabilities difficult.

Similarly, ways of automating the translation of the probability data (i.e., 72% good and 28% bad) into a DRIFT severity score will need to be explored.

Variables and their states relevant to Environmental Flow Assessments

Bayesian networks allow the use of various states, e.g., good and bad, or fast, slow, very slow to classify variables of interest. Although it is possible to include any number of states, it is advisable to limit these to two or three as the addition of each new state results in a geometric increase in complication of the resultant probability tables. We found that, when considering a species, the best results were achieved if we used the states Optimal, Sub-optimal and Lethal (e.g., as used in Figure 2). This allows the specialist to assess the suitability of any particular variable using tolerance data for the species of interest.

Threshold values

We incorporated threshold (or lethal) values into the Bayesian states to account for situations where change in one variable is such that the degree of change in other variables is irrelevant. For instance, in Figure 3, if salinities increase beyond a certain point then the minnow would not

be able to survive, whether or not riffle area was optimal. This is already done by the specialists in DRIFT but can be parameterized explicitly in Bayesian networks.

The effect of adding variables, and the need for standardization

In Figure 3, two variables were used and the resultant consequence for minnows was 72 percent good. In Figure 4, however, three variables were used (the first two with the identical values to those in Figure 3, and the third one being neutral) and the resultant consequence for minnows was 82 percent good. This dilution of a bad outcome as variables are added to the analysis is inherent in the calculation of probabilities and will be exacerbated by the addition of variables that are robust to changes in flow. Thus, the variables used would need to be chosen with care.

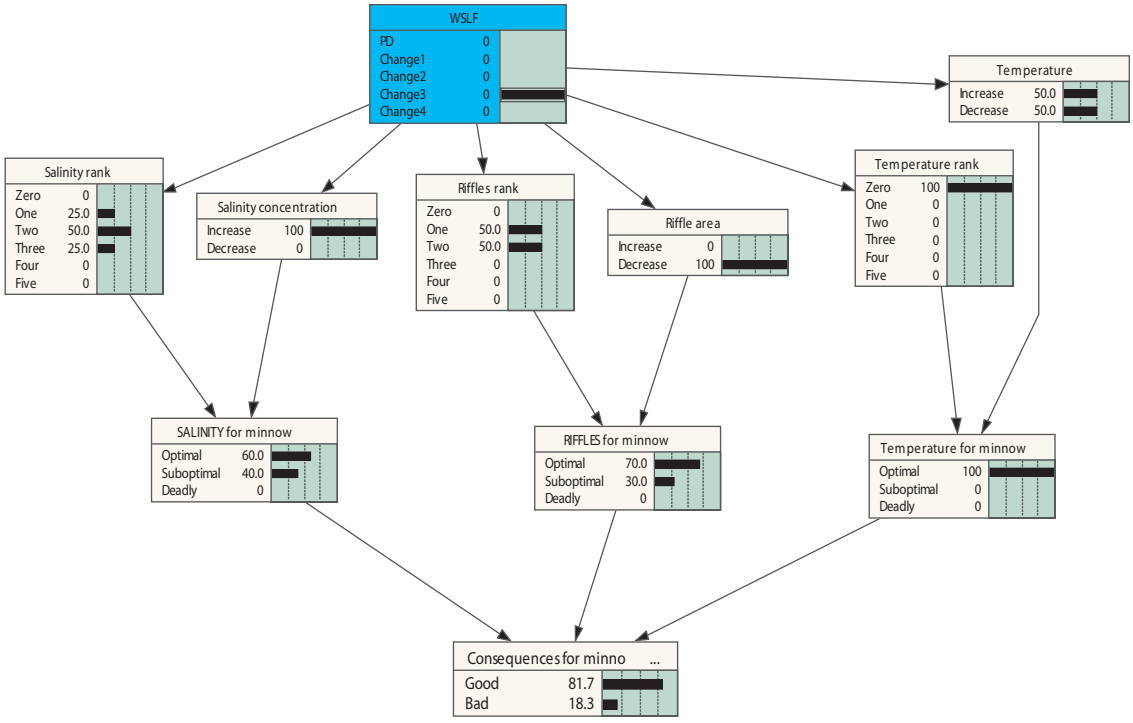
It may be necessary to set up a standard, limited set of variables applicable to a particular ecosystem sub-component that must be considered by the specialist rather than leaving the selection up to each individual specialist. Such a standard, limited set could be compiled from the scientific literature.

Furthermore, specialists will have to parameterize a tolerance box for every sub-component they choose to work with. Thus, it may make sense to set a minimum and maximum number of sub-components for each ecosystem component. It should make little difference to the overall structure of the Bayesian module whether the specialist chooses to model the species as a whole or to model the life-history stages of that species separately.

Building an information base

The variable-specific nodes generated for each ecosystem sub-component are not necessarily river specific and therefore could be transferred to other systems. Ultimately it should be possible to store variable-specific nodes created for one river on a website for adaptation and use in rivers where the same or similar sub-component occurs.

FIGURE 4.
Hypothetical Bayesian network constructed for DRIFT consequences of changes in wet season low flows for minnows, using three 'driving' variables, salinity, riffle area and temperature.



Conceptual relationship between DRIFT and Bayesian networks

Networks similar to those described above will need to be constructed for every species, guild and flow component of interest. However, this need not be done for all ecosystem components at once. It is possible to build the relationships in a modular fashion and to use a hybrid between a DRIFT and a Bayesian approach, concentrating on important components of the ecosystem, for instance fish. More and more Bayesian modules can gradually be added as they are developed.

Furthermore, retaining the original DRIFTSOLVER framework would make it possible to choose between a Bayesian network and other models (e.g., age-structured models) to assist in determining DRIFT severities, should they be available.

Eventually, a generic system of Bayesian modules could be constructed that includes all the variables and interlinkages for a particular type of river. The user would then be able to select the ones to be used, and have others taken out of consideration. A procedure (Netica) allows for 'node absorption', which means that a node can be removed from the Bayesian network without having to re-parameterize the probability tables.

Incorporating consideration of longitudinal interactions/basin level concerns

Bayesian networks will allow for inclusion of longitudinal biological interactions. Longitudinal issues relevant to each sub-component, such as whether or not a species needs to migrate, to where, how, at what life stages and time of year, etc., and the potential threats to these could be identified and included in the set of variables considered for that sub-component. These could then be dealt with according to the optimal, sub-optimal and lethal statements described above.

Recommendations

- Retain the DRIFT optimization framework for dealing with consequence data in DRIFT and DRIFTSOLVER.
- Retain existing scoring procedures for consequences of flow change.
- Explore the use of conceptual models and Bayesian networks in a modular fashion to assist with determination of raw consequences for key flow and ecosystem sub-components.
- Use the DRIFT lists of sub-components to guide development of conceptual models and the Bayesian framework.
- Explore ways to ensure that all weightings are explicit and can be recorded at all levels.

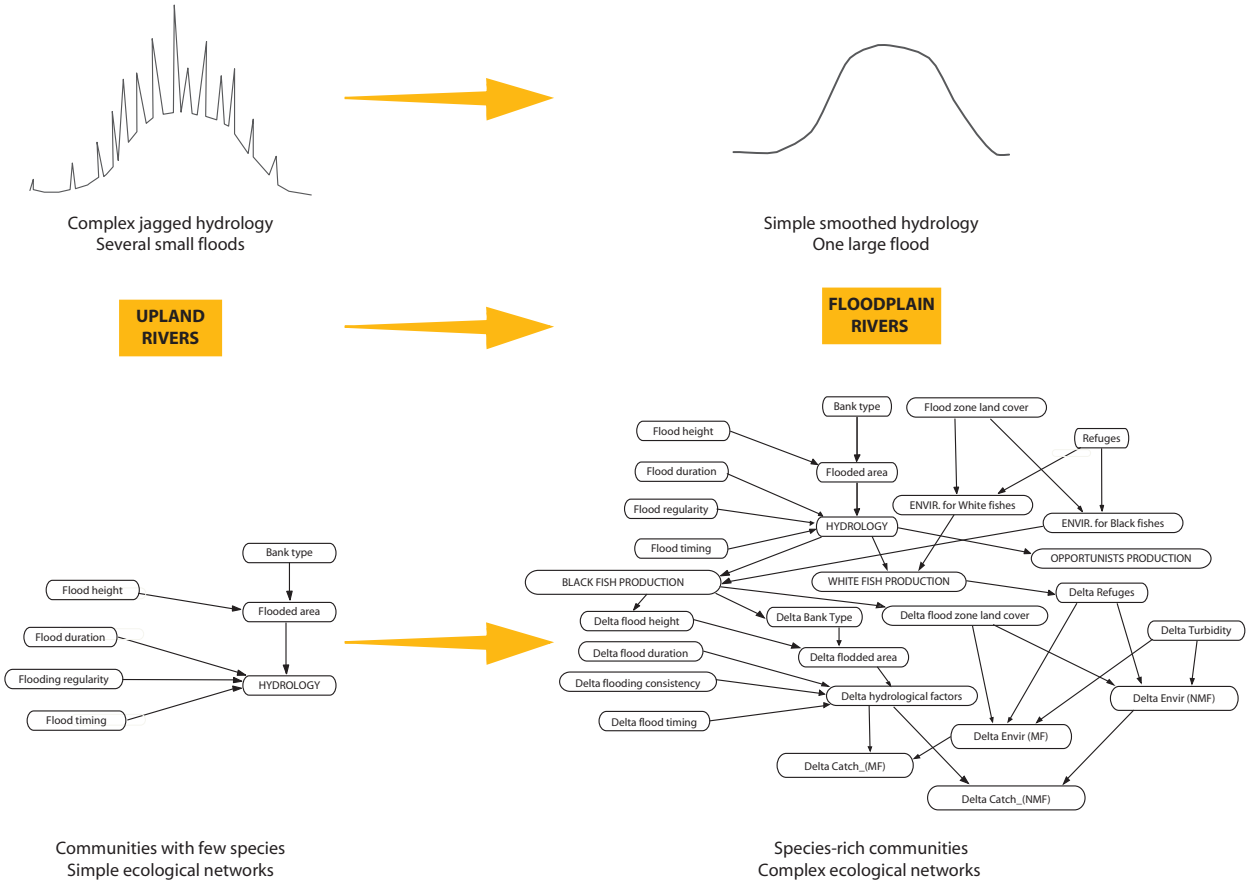
Adapting DRIFT for Large Rivers

DRIFT was originally designed for application in the rivers of southern Africa, which, with a few noticeable exceptions (e.g., Zambezi River, Mozambique), do not display the vast floodplains characteristic of some other areas of the world. Thus, there are several areas where DRIFT will require adjustment before it can be used successfully in floodplain rivers. These include, but are not necessarily limited to, terminology, hydrological classification, hydrological time steps and the content of the generic lists. These are dealt with in more detail below.

Linking DRIFT to a Bayesian network is possible and potentially beneficial. There will however be considerable variation between the details of how this is done for upland, mid and floodplain rivers.

We envisage that a DRIFT-Bayesian framework will be developed for each different river reach. The existing DRIFT (King et al. 2003; plus Bayesian networks) could provide the basis for upland rivers, and new frameworks, with time and spatial scales more appropriate for mid- and lower rivers, could be developed as required (Figure 5). For instance, there is some indication that it may not be necessary to look spatially at habitat in large rivers for fisheries alone although this will still be needed for assessment of fish biodiversity issues.

FIGURE 5.
Basic differences between upland streams and large floodplain rivers.



Proposed procedure for dividing a river basin into Environmental Flow Assessment zones

Changing flow classes, generic lists and time steps are seen as calibration details and should not necessitate changes to any overall DRIFT/ Bayesian framework. However, it will be important for any environmental flow study that the flow classes, time steps, and components and sub-components are compatible with those selected for other reaches of the study river.

The following procedure for dividing the river into reaches for Environmental Flow assessment is proposed.

For a study river:

1. Start with the hydrology and analyze variability of the flow regime with distance downstream of the headwaters, and the influence of major tributaries.
2. Choose hydrological zones representative of each level of variability.
3. Describe the ecological characteristics of the hydrological zones.
4. Determine a reach-specific division of the hydrological regime into flow classes based on the resolution needed for both meaningful ecological assessment and the sorts of management questions that will be asked.

5. Use the identified management questions to determine the level of changes of interest for each flow class, and the type of change (e.g., changes in frequency, variability, duration, and/or magnitude).

6. Back-check to ensure that the divisions make hydrological sense in terms of flow routing through the basin and in terms of data availability.

7. Choose a representative site(s) in each of the hydrological reaches of concern for the implementation of DRIFT.

This sequence is virtually identical to the process used in setting up a river basin for application of the Benchmarking Methodology (Brizga et al. 2002).

River Fisheries Models

Decisions by fishery managers on how to meet the various ecological, social and economic goals of a fishery need to be based on information. In the past such information was derived from simple scientific studies and took into account tradition, customs and taboos, as well as local and national political needs. More recent approaches to inland fisheries management need to deal with interactions with other users of the waters as well as purely fishery issues. As a result the need for scientific advice has intensified and now involves the use of one or more formal quantitative models of the fishery and the wider environment. Such models can be broadly categorized as empirical, population dynamics, or holistic.

Empirical Models

Empirical models are statistical representations of relationships between variables that do not generally refer to underlying processes. Empirical models may be fitted to observations from different river systems, made either at the same time or different times, or fitted to observations from the same river system but from different times. These models are typically fitted using linear (least-squares) or non-linear regression methods after appropriate data transformations to meet normality assumptions.

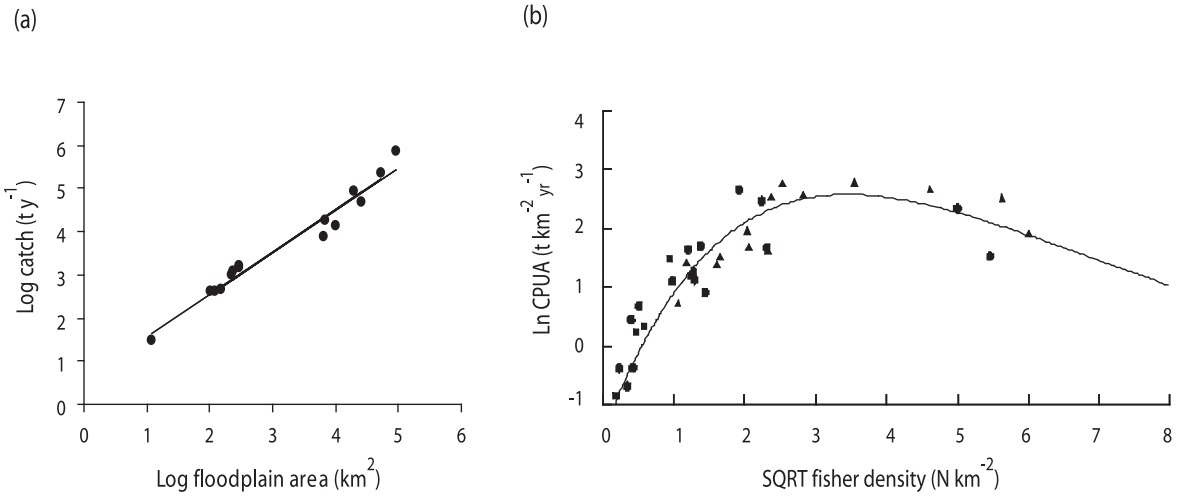
Empirical models have been used to describe the response of fish yield to one or more explanatory variables including measures of river morphology such as drainage basin or floodplain area (Lae 1992a, 1992b; Welcomme 1985), morpho-edaphic indices (Bayley 1988; Pusey et al. 1995, 1998, 2000), flow variables (Pusey et al. 2004) and fishing intensity (Welcomme 1985; Bayley 1988; Halls et al. n.d.) (Figure 6). They have also been developed to describe the response of fish yield to the quantity of freshwater discharged into estuaries (Loneragan and Bunn 1999).

A number of linear regression models of the general form of equation (1) have been used by numerous workers, including Welcomme (1975), Muncy (1978) and Van Zalinge (2003), to describe the response of fish yield in year *y* to some hydrological index (*HI*) in the same year or several previous years depending upon the age structure of the catch.

$$Yield\ (Y_y) = a + \sum_{i=0}^{n} b_i\ (HI_{y-i}) + \varepsilon_i \qquad (1)$$

The hydrological indices (*HI*) are chosen to capture key features of the hydrological regime that are believed to have a significant effect on yield, including maximum flooded area, depth, discharge rate, or functions of summed weekly water depths or areas during the flood and dry seasons (Figure 7).

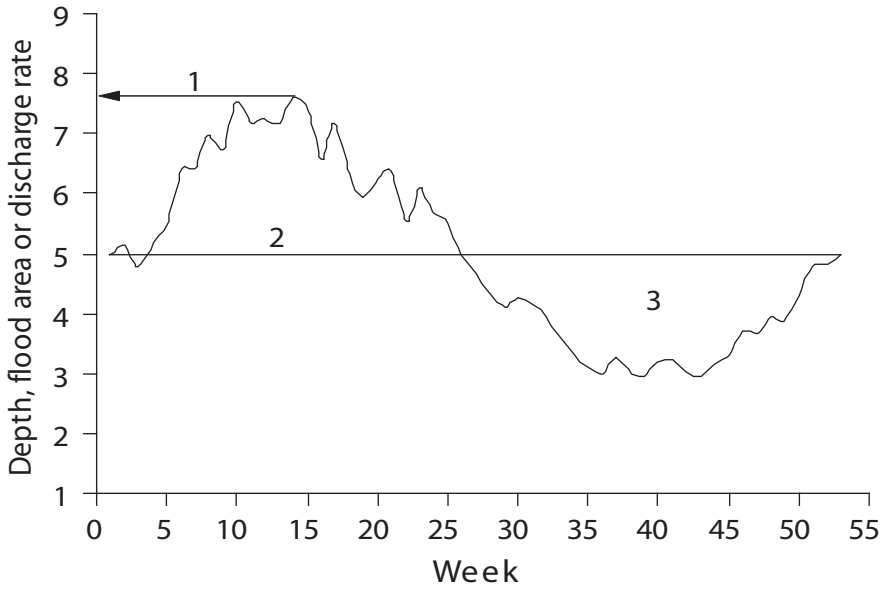
FIGURE 6(a).
A simple linear model describing the relationship between log-transformed floodplain area and catch; Figure 6(b). A modified Fox (Fox 1970) surplus production model of the form In *Yield = i* $^{0.5}$ exp (*a+bi* $^{0.5}$) + c describing the relationship between $\log_e$ transformed estimates of catch per unit area (CPUA) and square-root transformed fisher density estimates (*i*) for river systems in Africa (●); Asia (▲); and Latin America (■) where *a, b,* and *c* are fitted parameters.



Source: Modified from Welcomme 1985

Source: Halls et al. 2006

FIGURE 7.
Features of the hydrological regime commonly employed to construct hydrological indexes. Maximum flooded area, depth or discharge rate (1). Integrals of flooded area, depth or discharge rate based upon the flood (2) and dry season (3) periods.



Empirical models are easy to construct and are often robust. However, they rely on crude indices to describe complex hydrological conditions. In some cases, two distinctly dissimilar hydrographs can have the same flood index. Existing models fail to simultaneously account for variation in both hydrological conditions and exploitation intensity, despite the fact that both factors have been reported as significant in determining fish yield (Bayley 1988; Welcomme 1985, 2001). Moreover, because of the scale over which observations are made (typically the entire floodplain or drainage basin area of a river system), models of this type are often prone to significant measurement error. Consequently, model predictions have wide confidence intervals that significantly diminish their utility for decision-making. Adding to this is the unrealistic assumption that fish populations reach an equilibrium (stabilized) state within the model observation interval, usually one year, in response to any changes in hydrology and/or exploitation intensity. As a result, empirical models have mainly explored general relationships in river fisheries and can only be used for decision-making at the most generalized level. The statistical weakness of such models significantly limits their utility for supporting decision-making on the allocation of water, particularly for systems that exhibit significant variation in hydrological conditions and exploitation intensity, when decisions may have to be regularly revised or updated on an annual or multi-annual basis.

Population Dynamics Models

Fish population dynamics models attempt to describe the response of fish populations to exploitation and environmental variation based upon established theories of population regulation and recent advances in understanding of floodplain-river fisheries ecology and biology. These powerful *age-structured* or *biomass dynamics models* are capable of providing detailed insights into the way fish populations respond to simultaneous changes in both hydrology and exploitation through time and potentially, space. All the models assume that population biomass responds in some compensatory manner to changes in the hydrological conditions and to exploitation. In other words, under favorable hydrological conditions, exploitable biomass increases in response to improved opportunities for growth, survival and reproduction, but will decrease under less favorable hydrological conditions. The main differences among the proposed models lie in the way the underlying processes regulating populations are assumed to operate, in the way they are modeled, and the techniques employed to estimate model parameters.

Biomass dynamics models

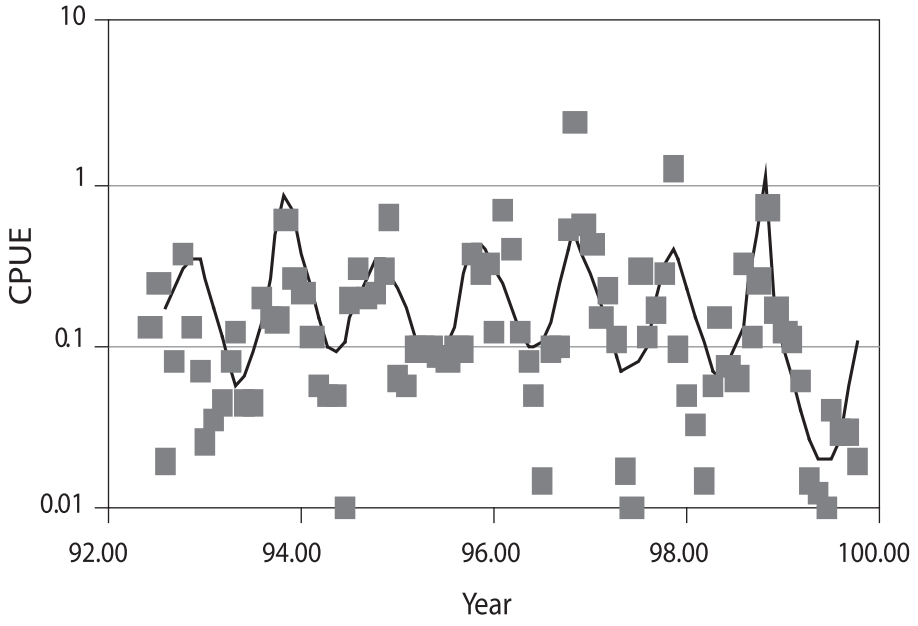
Biomass dynamics models (Hilborn and Walters 1992) treat the whole fish stock as a pool of biomass subject to production, that is, the net result of growth, reproduction and natural mortality, and harvesting.

The processes of growth, mortality, and recruitment determining exploiting biomass in the extended Fox biomass dynamics model, are described by a single parameter *r,* the intrinsic rate of population growth. However, in this extended form, hydrological variation (measured in terms of flooded area) is assumed to affect both the environmental carrying capacity of the habitat, *K,* and the catchability coefficient, *q* (through variation in the density of the population). These flooded area effects are modeled as power functions of water area, *a* expressed as a proportion of an arbitrarily defined reference area, A (e.g., the mean dry season flooded area). The resulting model is:

$$B_{t+1} = B_t + rB_t \left(1 - \frac{\log (B_t)}{\log \left(K \left(\frac{a_t}{A} \right)^p \right)} \right) - C_t \qquad (2)$$

$$C_t = B_t E_t q \left(\frac{a_t}{A} \right) \qquad (3)$$

FIGURE 8.
Observed and Predicted CPUE (catch per unit effort) for the spotfin swamp barb *Puntius sophore* in Gotokbari beel, Bangladesh.



Source: Lorenzen et al. 2002

where B_t is the biomass at time *t,* C_t is catch, a_t is the water area at time *t* relative to a reference area *A, p* is the scaling factor of ecological carrying capacity with area, and *c* is the scaling factor of catchability with area. The model has been successfully fitted to daily time series of catch, effort and water area data for a Bangladesh beel fishery (Figure 8). Although it is fitted to a single species here it could equally well be fitted to a complete species assemblage.

Minte-Vera (2003) describes a version of the discrete lagged-recruitment, survival and growth (LRSG) model (Hilborn and Mangel 1997) where the biomass at time t+1 is expressed as:

$$B_{t+1} = B_t s + R_t - C_t \qquad (4)$$

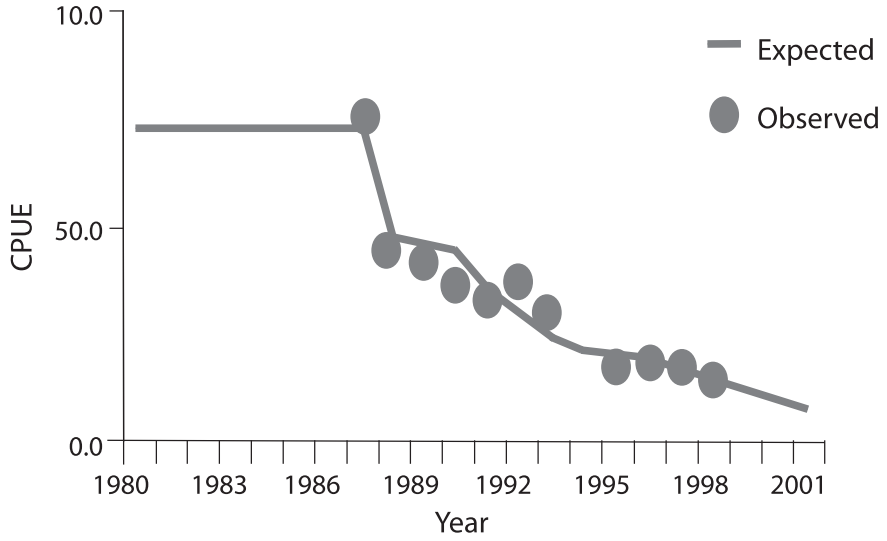
where B_t is the biomass at time *t, s* is the rate of population biomass growth arising from somatic growth and survival, R_t is the recruitment (juvenile biomass) in time *t* and C_t is the catch during time *t.* Recruitment is described by an extended form of the Beverton and Holt stock-recruitment relationship (SRR) with a lag of two years between recruitment and spawning biomass and an extra term, *c* to describe the effect of relative flood strength on recruitment success (equation (5)). Here d_{t-2} is the number of days flooding above some arbitrary height in year *t-2,* $\bar{d}$ is the average number of days flooding above this arbitrary height and α and β are the parameters of the basic SRR.

$$R_t = \frac{B_{t-2}}{\alpha + \beta\, B_{t-2}}\, exp(c\,(d_{t-2} - \bar{d})) \qquad (5)$$

Catches of *Prochilodus* from the Parana River Basin in Brazil have been successfully predicted using this delay-difference model (Figure 9). Minte-Vera notes, however, that the model is currently not capturing the effect of the variation in flood strength on recruitment. This she postulates may reflect the inadequacy of a deterministic recruitment model to describe the intrinsically variable (stochastic) nature of recruitment in this species.

Both the Lorenzen and Minte-Vera models need fairly extensive catch-effort datasets to provide satisfactory answers.

FIGURE 9.
Observed and predicted *Prochilodus* CPUE (catch per unit effort) from the Parana River Basin in Brazil.

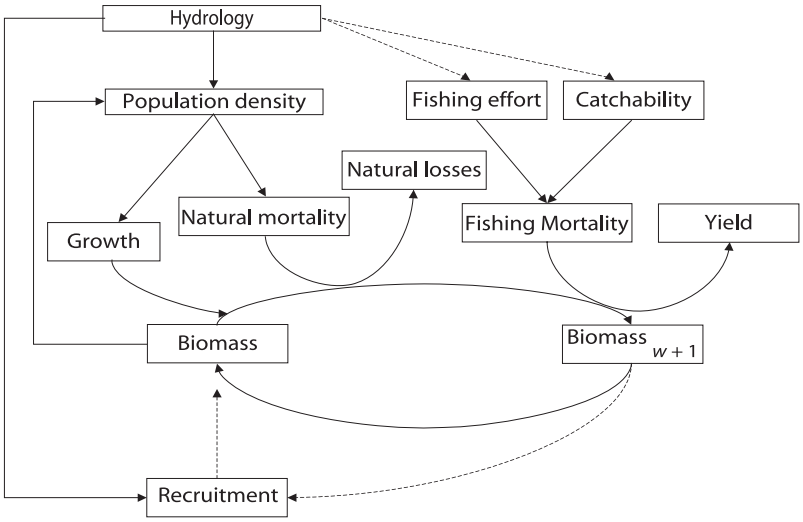


Source: Minte-Vera 2003

Age-structured models

Age-structured or *dynamic pool models* attempt to account explicitly for the processes of growth, mortality and reproduction to predict changes in number and biomass in response to exploitation and environmental variation (Figure 10). Whilst more complex, these models are more realistic than biomass dynamics models since they can take better account of the various time-delays and changes in population structure associated with growth and recruitment and, therefore, are likely to provide more precise predictions under conditions of rapid population change (Hilborn and Walters 1992) typically exhibited by floodplain fish

FIGURE 10.
Schematic representation of the population model illustrating the processes by which the biomass in week *w* becomes the biomass in the following week, *w*+1. The weekly process is repeated for the 52 weeks of the year, after which recruitment, determined by the surviving spawning stock biomass, is added at the end of week 52. Solid lines indicate direct influences or operations and broken lines indirect influences or occasional operations.



Source: adapted from Welcomme and Hagborg 1977

populations. Their transparent nature also facilitates improved understanding of the response of populations to external factors and the exploration of a greater range of potentially effective management interventions.

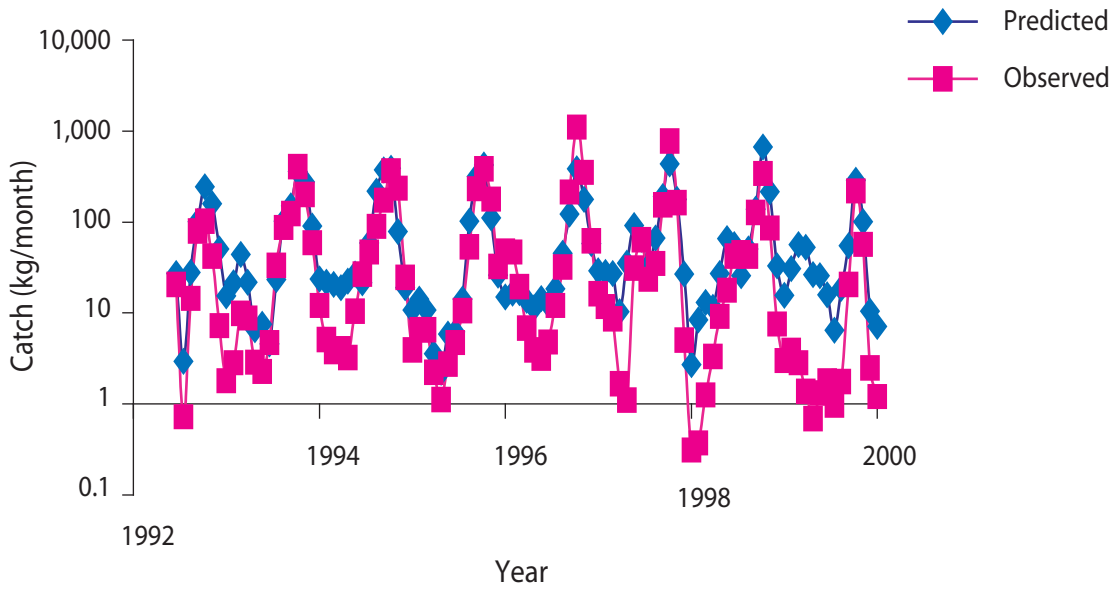
Age-structured models were first applied to floodplain river fisheries in 1977 to explore the dynamics of African fish populations under different regimes of flooding and exploitation (Welcomme and Hagborg 1977). They were applied later to the lakes of Madagascar (Moreau 1980), and the Central Delta of the Niger (Morand and Bousquet 1994).

The age-structured model described by Halls et al. (2001) builds on this earlier work, incorporating more conventional sub-models describing the (density-dependent) processes of growth, mortality and recruitment that exploit new insights into the dynamics of floodplain fish populations gained during the last two decades. Growth, via the asymptotic length, is modeled to decline linearly with increasing biomass density reflecting increased competition for food resources. Recruitment is described by an extended Ricker stock-recruitment relationship with an extra term to describe the effect of changes in system fertility with flood strength on recruitment success. Mortality is modeled to increase linearly with increasing numerical density reflecting numerical and functional responses by predators and declining water quality, and competition for shelter from predators. Catches are also modeled to be density-dependent, through the effect of biomass density of gear catchability.

The model is general and flexible and can be easily modified to include a range of other species with or without interaction. Annual yield or exploitable biomass can be predicted with the model allowing for weekly variations in age-dependent growth and mortality rates and inter-annual variations in recruitment strength driven by changes in population density in response to exploitation and/or dynamic hydrological conditions (Figure 11). The model algorithms are given in Halls et al. (2001) and Halls and Welcomme (2004).

Once established the model does not need extended datasets as estimates for many of the model parameters can be drawn from the literature or FishBase. Those that cannot be so derived can be estimated from empirical relationships derived from spatial comparisons or experimentation within a single year (e.g., Halls et al. 2000) or from among-population comparisons (e.g., Lorenzen and Enberg 2002). Using key model parameters derived from experimentation

FIGURE 11.
Observed and predicted catches for the spotfin swamp barb *Puntius sophore* in Gotokbari beel, Bangladesh.



during a single year, the model has successfully predicted the same time series of catches from the Bangladesh *Puntius* fishery described above on the basis of weekly estimates of flooded area and fishing effort (Figure 11).

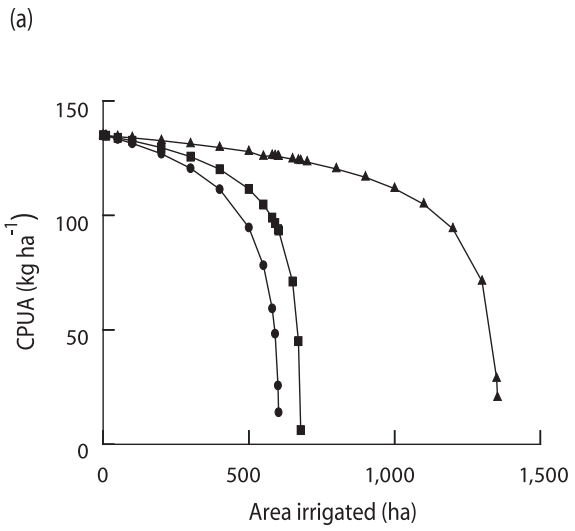
These population dynamics models provide a powerful means to aid decision-making with respect to the integrated management of water, fisheries and other water-dependent resources such as agriculture. For example, the age-structured dynamic pool model described by Halls et al. (2001) has been used to quantify the impacts of modified hydrological regimes on fish production inside flood control compartments in Bangladesh and to identify mitigating measures including the retention of more water during the dry and closed seasons (see Halls et al. 1998, 1999). Shankar et al. (2002, 2003) use the model to explore economic tradeoffs between dry season crop irrigation and fisheries in Bangladeshi floodplains (Figure 12a). Similar water allocation decisions can be guided with outputs generated by biomass dynamics models (Figure 12b).

These types of models can also provide inter and intra-annual predictions of the effects of modifications to the natural hydrological regime of rivers on exploitable fish biomass. Halls and Welcomme (2004) illustrate how the model can be used to generate guidelines for managing and manipulating hydrological conditions in river systems including the release of artificial floods downstream of dams and the manipulation of water levels within impounded floodplains, either to minimize the risk of species extinction (stock collapse) taking account of existing and planned patterns of exploitation, or to maximize exploitable biomass (Figure 13).

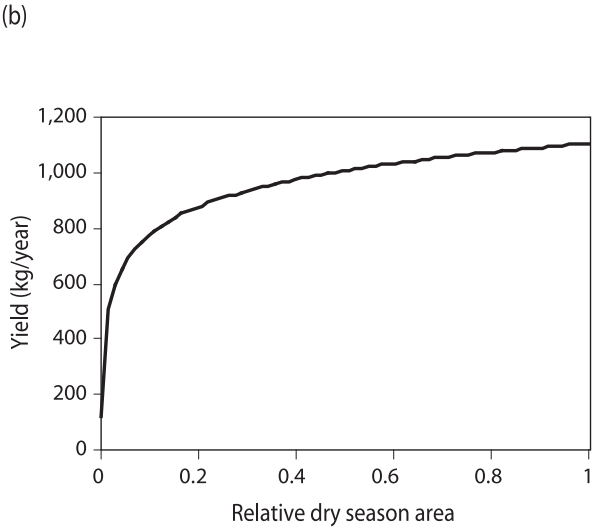
Holistic Fisheries Models

In the field of environmental flows assessment the term "holistic methods" originates from the work of Arthington et al. (1992, 1998, 2004b) and King and Louw (1998), through several developments and applications in Australia and South Africa. In the field of inland and floodplain fisheries, the term

FIGURE 12a.
The effect of dry season irrigation for boro rice on fish production in Bangladeshi floodplains measured in terms of catch per unit area (CPUA). Figure 12b. Predicted yield of *P. sophore* as a function of dry season area relative to flood season area.

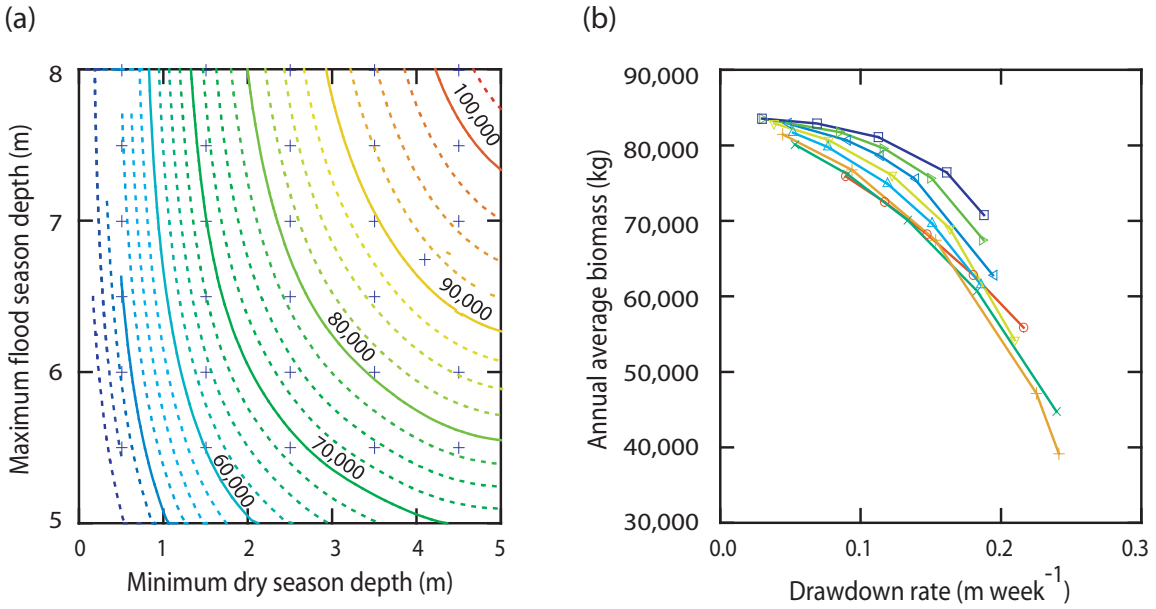


Source: Shanker et al. 2003



Source: Lorenzen et al. 2002

FIGURE 13.
Examples of the types of outputs that can be readily generated with age-structured models illustrating (a) exploitable biomass as a function of flood and dry season channel depth for a 25 week flood season duration (figures on lines = total exploitable biomass (kg)); and (b) exploitable biomass as a function of drawdown rate for flood season durations ranging from 5 to 40 weeks (lines refer to flood duration in weeks: ○ = 5; x = 10; + = 15; △= 20; ▽= 25; ◁= 30; ▷= 35; □ = 40).



Source: Halls and Welcomme 2004

applies to models that address fish production or yield in the broader context of environmental management, and therefore integrate a diversity of variables of hydrological, environmental or social nature (Lorenzen et al. 2007). The earliest example of a fully holistic fisheries model is that of the fisheries in the Central Delta of the Niger River (Bousquet 1994b), that calculates fish catch while integrating the strategies and requirements of individual fish species, as well as river hydrology, floodplain morphology, fishing effort and adaptive behavior of fishers.

Holistic models can be broadly classified into ecological models, multi-agent models and Bayesian networks.

Ecological models

Ecological modeling aims at modeling the whole ecological network in a given ecosystem, from primary production to top predators, including fishers. The dominant modeling framework is Ecopath with Ecosim, with which 44 site-specific models have been developed so far in freshwater systems, including 25 in Africa (see www.ecopath.org). The principle of Ecopath (Christensen and Pauly 1992, 1993; Walters et al. 1997) is to create a mass-balanced model of the ecosystem, represented by trophically linked biomass units that consist of single species, ecological guilds or life history stages. The model is based on two core equations:

(i) an equation describing the production term for each unit:

Production = catch + predation + net migration + biomass accumulation + other mortality

(ii) an equation of conservation of matter within a unit:

Consumption = production + respiration + unassimilated food

In general, an Ecopath model requires three of the following parameters for each unit: biomass, production/biomass ratio (or total mortality), consumption/biomass ratio, and ecotrophic efficiency.

Although popular for enclosed or well-studied large aquatic ecosystems, this approach is data and knowledge hungry and requires quantitative information on biomass of certain groups such as phytoplankton, zooplankton, benthos, and on related trophic flows. Such data are simply nonexistent for most tropical rivers and floodplains, hence the absence of these systems among the applications of Ecopath with Ecosim.

Multi-agent models

In a fishery the fishers are subject to the spatial, temporal and biological variability of the resource, and respond by adopting a range of different exploitation strategies. However, the variable exploitation strategies and more generally the interactions between the human actors and the resource are rarely taken into consideration by fisheries modeling approaches. For instance, as noted by Weisbuch and Duchateau-Nguyen (1998), *"although fisheries have been modeled by economists from an integrated perspective since the 30s, the influence of cultural constraints on resource exploitation is seldom taken into account".*

It is only recently that a field of modeling has started to address social processes and their dynamic links with the natural environment. This approach involves a coherent conceptual basis (e.g., Ostrom et al. 1994; Holling 2001; Janssen 2002) and relies, for modeling, on the tools of Distributed Artificial Intelligence (Ferber 1999). Among the underlying principles are: (a) that variability is an inherent component of natural but also social systems; (b) that addressing issues of scale and scale transfer is essential; (c) that there are as many viewpoints on the dynamics of a system as there are stakeholders; (d) that a diversity of individual behaviors often results in an emerging process that is different from the sum of individual behaviors.

The modeling of human-environment interactions has been based mainly on multi-agent systems. Agents are programmable entities able, in a limited way, to sense and react to their context, to communicate with one another, to accept and set themselves goals and to maintain and update individual belief sets (after Doran 1997). A summary of this modeling approach is given in Bousquet et al. (1999).

Although this school of modeling is rapidly expanding, agent-based modeling has addressed floodplain fisheries in one instance only (Bousquet 1994a, 1994b). This model of the fisheries of the Central Delta of the Niger River has several components:

- a hydro-ecological one, with four spatial sub-components: river mainstream, channels, floodplain and ponds and their attributes (flood threshold, water level, food provided, connected biota);
- a fish component, made of major species and their attributes (growth, diet, fecundity, migrations, natural mortality, biotope used, etc.); and
- a social component and its attributes (access to each water body, gears, ethnic group, asset, etc.).

These components are connected according to descriptions given by a multi-disciplinary team of experts, including hydrologists, biologists, sociologists and anthropologists. Fishing behavior, for instance, depends on the ethnic group of the fisher: "if fisher belongs to ethnic group X, then access to biotope Y is forbidden in season Z"; therefore the total proportion of fishers of ethnic group X will influence the mortality of targeted species S_1, S_2 and S_3 that use biotope Y in season Z. In this model the agents have a capacity of learning: for instance if in year *n* the average catch of fishers using gear G_1 is higher that that of fishers using gear G_2, then in year *n*+1 the proportion of fishers using gear G_1 will be higher (which results in turn in higher mortality rates on species targeted by gear G_1). The approach is detailed in Bousquet et al. (1993) and Cambier et al. (1993).

Multi-agent systems are the most holistic modeling approach being developed for the management of human influenced natural systems (e.g., FIRMA 2000[1]; see also http://cormas.cirad.fr/indexeng.htm). They are also underpinned by a strong theoretical background that reflects the complexity of the real world much more precisely than equation-based models. However, they require a strong background in computer science and, although they are becoming increasingly popular, their complexity ultimately requires that the modeling team is fully trusted by the decision-makers if the model outputs are to be converted into management decisions.

Bayesian networks

The need to make decisions in uncertain conditions led to the use of probability theory in the development of modeling tools. In the field of fisheries this has resulted in a large number of applications (see Punt and Hilborn 2001 for a review), but until very recently (Minte-Vera 2003) models of inland fisheries have not been developed that incorporate probabilistic components. Besides this refined approach to classical fisheries modeling, a new development consists in the combination of probability theory (in particular the Bayes' theorem for probabilistic inference, Bayes 1763) with the theory of graphs (Jordan 1999), to result in a graphic modeling approach called Bayesian networks. Although this technique is applied here primarily to fisheries, it can be extended to include all other components of the river-floodplain system and is thus a holistic method.

The essence of Bayesian networks consists in defining the system studied as a network of variables linked by probabilistic interactions (Jensen 1996). Bayesian networks (also called Bayes nets or Bayesian belief networks) were developed in the mid-1990s as Decision Support Systems for medical diagnostic and financial risk assessment. A detailed but accessible description of these methods can be found in Charniak (1991), and for more specific ecological and management applications, in Ellison (1996), Cain (2001) and Reckhow (2002).

- Bayesian networks are based on variables representing the modeled environment. Variables can be quantitative (e.g., "Flood level") or qualitative (e.g., "Fishing strategy"). For each variable a small number of classes is defined (e.g., "Flood level" can be characterized by a few numerical classes such as 0 1m, 1-2m, 2-3m, or "Fish production" by qualitative classes such as Low, Medium and High).

- In the network, variables are connected by links expressed in terms of probabilities. This step is called elicitation of prior probabilities. "Prior probabilities" are those entered during the building phase of the model, as opposed to those calculated by the model and called "posterior probabilities". If data is available, then the quantified relationship between variables A and B is converted into probabilities. For instance studies of the relationship between flood level A and floodplain fish production B, expressed as B = f(A), can be used to define the chance of having a high fish production (Low, Medium, High) for a given flood level (0-1m, 1-2m, 2-3m). New ways to develop such relationships are outlined in Arthington et al. (2006). If data is not available, Bayesian networks also allow expert knowledge to be used to characterize the known relationship between two variables, which is again expressed in terms of probabilities. This possible integration of expert knowledge (i.e., knowledge from scientists, field practitioners or persons with specific experience of the topic addressed) into a

[1]FIRMA 2000 Freshwater Integrated Resource Management with Agents. A project supported by European Union ("Framework 5 Programme for Research and Development") and by the European Commission ("Key Action on Sustainable Management and Quality of Water programme") [see http://firma.cfpm.org]

modeling framework contributed significantly to the success of this approach, and explains the term "Bayesian belief networks, or BBN".

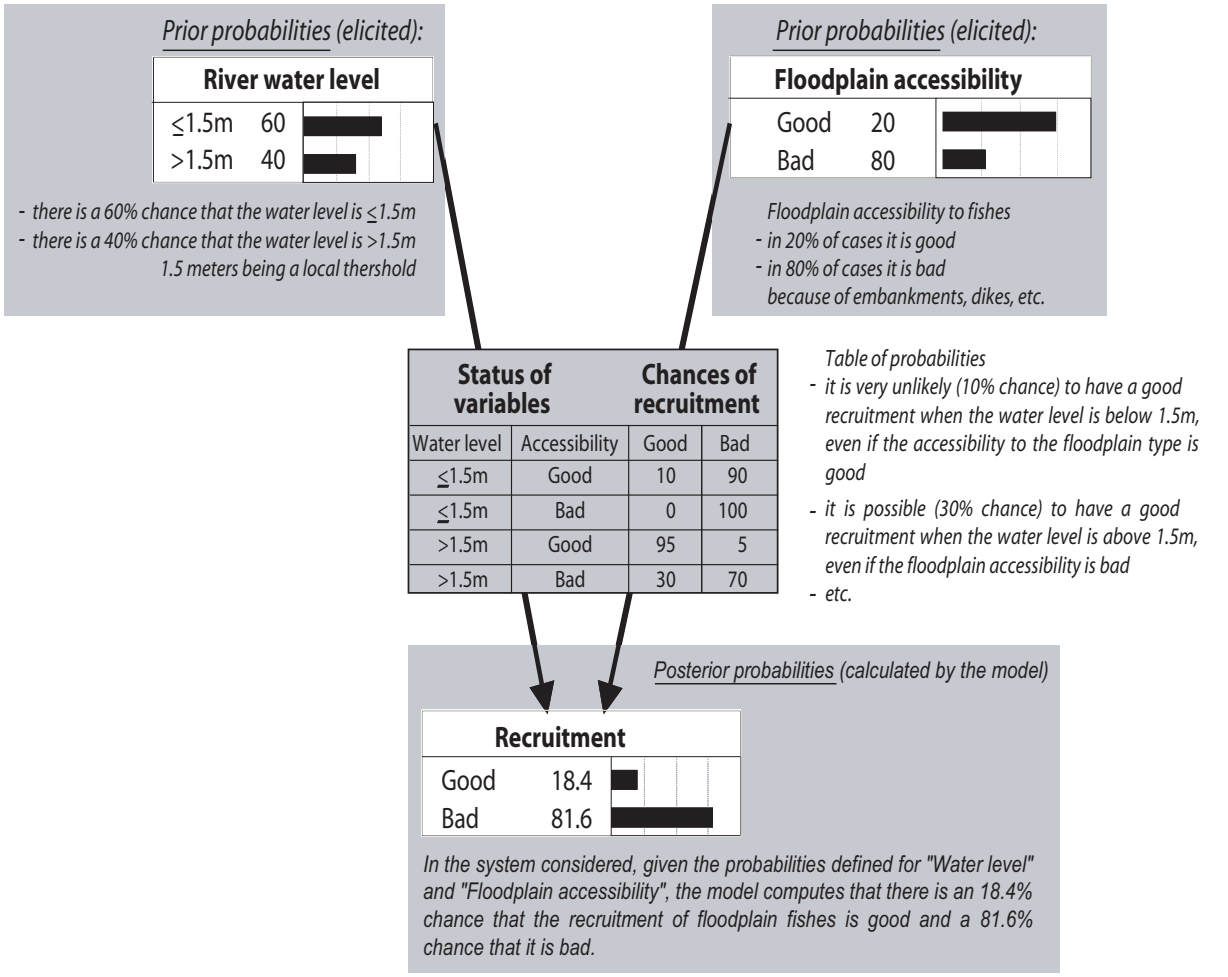
- Ultimately the model calculates, based upon the Bayes' formula, the overall trend resulting from the interaction of probabilities within the system. A Bayesian network is usually built in such a way that a diversity of influencing variables (e.g., water level, floodplain accessibility) converge towards the variable one is specifically interested in (e.g., fish recruitment).

An example of variables in a Bayesian network is shown in Figure 14.

In the field of environment, Bayesian networks were originally developed for integrated natural resources management (Varis et al. 1990; review of early works in Marcot et al. 2001) and became popular for integrated water management (e.g., Varis 1998; Batchelor and Cain 1999; Varis and Lahtela 2002; Borsuk et al. 2002b).

Bayesian networks have been used for inland fisheries management to assess risks of population extinction (Shepard et al. 1997; Lee and Rieman 1997), consequences of land use options (Rieman et al. 2001; Marcot et al. 2001), and environmental drivers of fish

FIGURE 14.
Three variables connected by probabilistic links in a Bayesian network (example not related to a particular case study). Networks can be made of several dozens of variables.



production (Baran and Cain 2001; Borsuk et al. 2002a; Baran et al. 2003). This latter paper is the only one dedicated to tropical floodplains (Figure 15). Fisheries management has also been addressed by Peterson and Evans (2003) and Kuikka et al. (1999).

The software used for these models is either commercial or developed by the researchers themselves. Comprehensive reviews of software are available on several web sites, including:

- http://www.ia.uned.es/~fjdiez/bayes/software.html

Bayesian networks offer a solution to the common problem of data scarcity by the possible use of expert knowledge; they are easy to compute, are intuitive and visually explicit, and thus are good tools for determining global trends and communicating summaries of complex information beyond the reach of individual experts or decision-makers.

The definition (or elicitation) of prior probabilities is the most delicate part of the modeling process, and its quality determines that of the whole model (e.g., Chen et al. 2000. The elicitation of probabilities in a Bayesian belief network ideally requires the contribution of a multidisciplinary group of specialists (Cain 2001; Borsuk et al. 2001), but defining modalities of this consultation in view of minimizing biases and maximizing representativity of the model is still a theme for research.

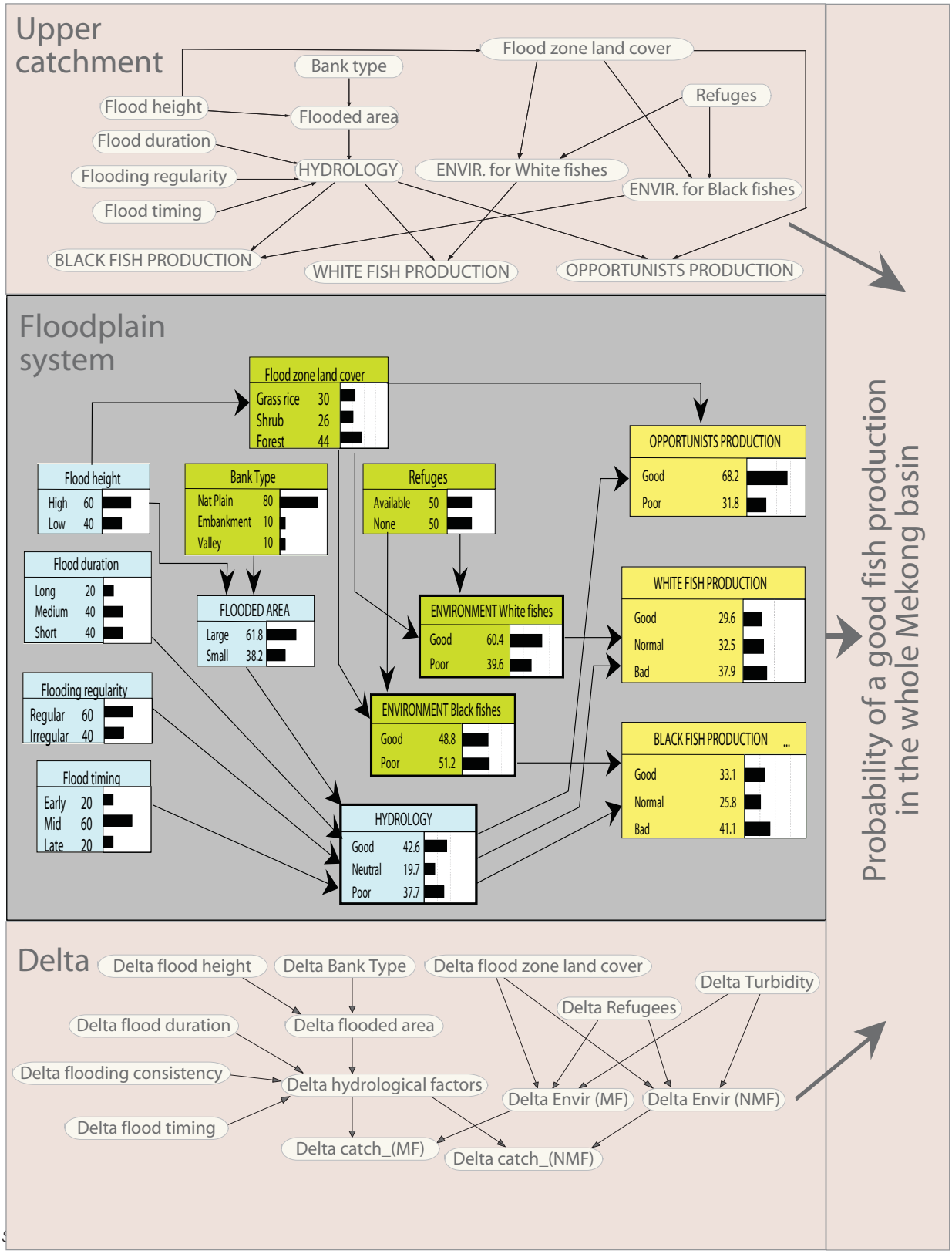
Another interesting feature of Bayesian networks is associated with the consultation process: the possible involvement of stakeholders in the building of the model. It allows the integration of various viewpoints on the system studied (those of the scientists, but also of managers, of farmers or fishers, etc.) and facilitates the appropriation of the model and the use of its outcomes by managers and decision-makers. Most authors have outlined the importance of the consultation process (e.g., Moss et al. 2000; Borsuk et al. 2001; Soncini-Sessa et al. 2002; Peterson and Evans 2003) and its role in bridging the common gap between the provision of scientific advice and its use in decision-making. In this field of application probably lies the strongest potential of Bayesian networks for inland fisheries management.

Dealing with Uncertainty

This assessment has so far described fisheries and environmental models and methods that provide advice for water management decision-making, including some approaches that use Bayesian methods. Uncertainty is inherent in all these models particularly with respect to how well they represent real systems and how precisely they provide parameter estimates. This uncertainty results from scarce or imprecise data and information, the existence of several hypotheses about how the system works and the sheer variability of natural ecosystems. Despite these uncertainties decisions concerning environmental flows and fisheries still need to be made. Management decisions must, therefore, explicitly account for these uncertainties by quantifying the risk of a particular course of action. Bayesian methods provide a powerful means to measure such risks. The theory behind these techniques is outlined below.

A Bayesian risk assessment integrates a model, such as the ones described above, with prior information and data collected for the ecosystem in order to estimate the probability of different states of the populations and of different values for the parameters (Hilborn and Mangel 1997; Punt and Hilborn 1997; Wade 2000). Bayesian methods have been used successfully with population dynamics models (see McAllister et al. 1994; Meyer and Millar 1999).

FIGURE 15.
Summary of the natural fish production model developed for the Mekong River Basin.

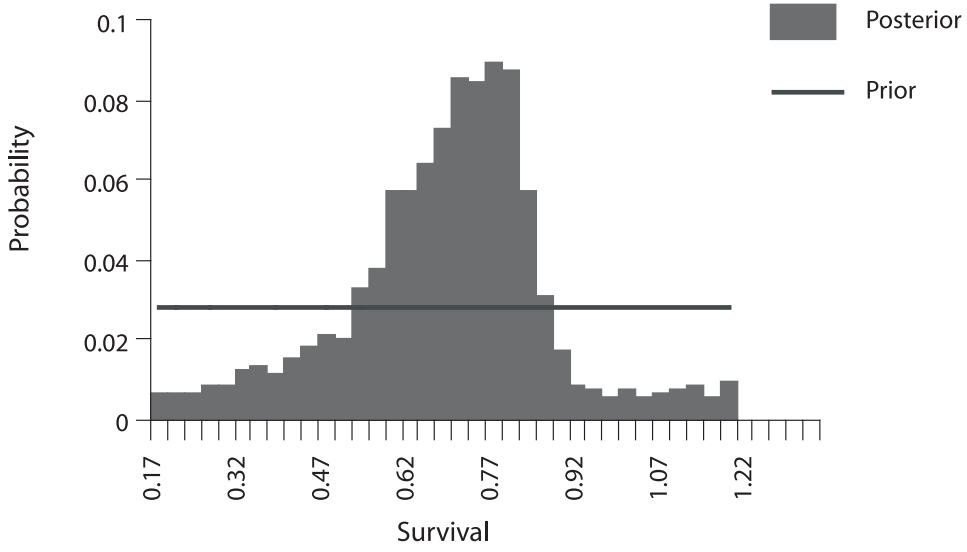


Model and their possible parameter values summarize the different hypotheses about an ecosystem. It is important that we know what the evidence is for each hypothesis. In a Bayesian analysis we would start by stating the evidence for each hypothesis in terms of a prior probability distribution. Then the data are used to update the prior distribution and obtain a posterior distribution. For example, in Figure 16 no knowledge existed about the parameter, all the possible values were equally likely. After the use of the data, a posterior probability was picked indicating that the data supported the hypotheses that the parameter is between 0.6 and 0.8.

The first step in a Bayesian analysis is to choose prior distributions for all the parameters used in the model. For most ecological problems we have prior information that is valuable and should not be wasted. This information can be reflected in an informative prior distribution that will reduce the uncertainty surrounding parameter estimates. Knowledge may come from scientific results (from studies in other systems, or on other species), expert opinions and traditional knowledge. For example, when using an age-structured model the values of the density-dependence parameter obtained experimentally for *Puntius* by Halls et al. (2001) can be used to derive a prior distribution when applying the same model to other species. A combined analysis of several datasets such as the Lorenzen and Enberg (2002) study on density dependency can also be used. Another way to derive a prior distribution is to use opinions from experts or users of the resource, who may have a good idea about what values are plausible. Users such as fishers have traditional knowledge transmitted between generations or acquired by repeated observation of the system. Another promising way to reduce uncertainty is to implement experiments designed to estimate the values of certain parameters that can be used as priors in models (Arthington and Pusey 2003; Poff et al. 2003).

The second step in a Bayesian analysis is to incorporate data for the system that is being studied. The degree to which the data support a hypothesis can be derived. For example, when using an age-structured model, the available data may be catch-per-unit-of-effort, abundance estimates and length-frequency data. These data are related to the population dynamics model using probability distributions. If the data are informative about the population, this will be reflected by the stronger selection of particular values in the posterior distribution than in the prior.

FIGURE 16.
Prior and posterior probability function for the growth-survival parameter for the Curimba in Parana River (Minte-Vera, 2003). The prior was uninformative and the posterior was driven by the data. The data supported the hypotheses that the parameter is between 0.6 and 0.8.



The results of the Bayesian estimation (posterior probability distributions) can be used in a risk assessment to calculate the risk of taking a particular course of action. A risk assessment is the evaluation of the consequences of alternative management action under uncertainty. A risk analysis involves six steps (Punt and Hilborn 1997; Punt and Hilborn 2001):

Step 1: Identifying the alternative hypotheses about the system;

Step 2: Determining the relative weight of evidence in support of each alternative hypothesis (expressed as a probability);

Step 3: Specifying the alternative management actions;

Step 4: Specifying a set of performance statistics to measure the consequences of each management action;

Step 5: Calculating the values of each performance statistic for each combination of a hypothesis and a management action; and

Step 6: Presenting the results to the decision makers.

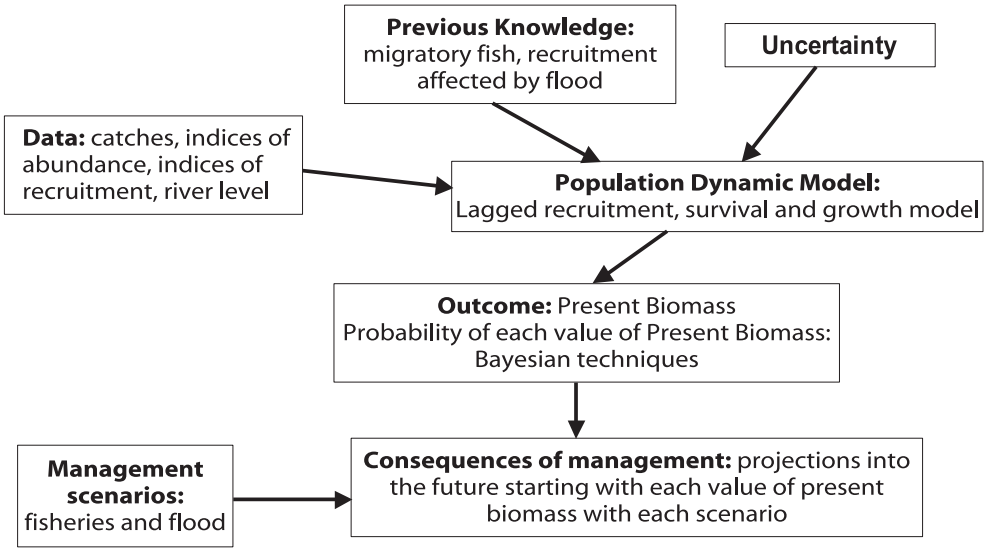
Steps 1 and 2 consist of integrating all prior information, data and models in a Bayesian estimation procedure as described above. The alternative management actions (step 3) are discussed with managers. For environmental flow assessments, management actions could include modifications in the flow regime, changes in harvest levels or changes in habitat. The performance statistics (step 4) should be chosen to quantify the management objectives. For example, population dynamics models can produce several performance statistics such as population size, the average body size of a fish, the average catch, the probability of falling below some threshold level, etc. Step 5 is performed by projecting the population into the future many times using values drawn from the posterior distribution of the parameters as starting points for each management scenario. The results of the risk analysis may be presented to decision makers (step 6) in the form of a table or a set of curves showing the probability of values of the performance statistic. For example, if the performance statistic is chosen to be the ratio between the population size in one year and the current population size, we could assess the probability of population increase or decrease given each management scenario. A conclusion such as "with decrease in 30 percent of the environmental flow, the population has 20 percent chances of decreasing by 50 percent or more" can be drawn from these outputs.

Minte-Vera (2003) applied the discrete lagged-recruitment, survival and growth (LRSG) model (Hilborn and Mangel 1997) described above in a risk assessment for the migratory Curimba *Prochilodus lineatus.* Figure 17 presents a schematic view of the risk assessment.

The management scenarios explored where different levels of allowed catches combined with different flood regimes.

The output of the assessment was presented as a decision table (Table 10). The decision table presents the probability of each possible outcome, in this case the current population size. For each of those population sizes as a starting value, the population was projected 4 years into the future applying the management option. The current population size is most likely between 150 to 500 tonnes (probability = 0.686). The performance statistic was the ratio between the population size in 2005 to the current population size. A value above one will indicate increase in population size and a value below one will indicate decrease. The decision table indicates that fishing is only sustainable at a level of 50 tonnes if the population size is above 700 tonnes. The probability of the population being above 700 tonnes is very low, indicating that it is very likely that fisheries are not sustainable. The regulation flow as modeled is not likely to affect the population as the fisheries are.

FIGURE 17.
Schematic view of the Bayesian-based risk assessment for the Curimba in Parana River, Brazil.



Source: Minte-Vera 2003

TABLE 10.
Decision table for the Bayesian-based risk assessment for the Curimba in Parana river (Brazil) showing short-term effects of the management options. The performance statistic is the ratio between the population size in 2005 to the population size in 2001. A ratio above 1 indicates increase in the population size. The average of the performance statistic is obtained by multiplying the ratio by the probability of each value of the population size in 2001.

			Population size in 2001 (biomass in tons)					
			<150	150 - 300	300 - 500	500 - 700	700 - 1000	1000 - 3000
			Probability of each population size					
Management Options			0.123	0.314	0.372	0.073	0.025	0.093
Flood	Fishing	Average performance	Performance statistic (biomass 2005/biomass 2001)					
Natural	No	1.12	0.32	1.23	1.25	1.22	1.29	1.10
	50 tons	0.82	0.12	0.80	0.95	1.01	1.17	1.05
	100 tons	0.54	0.03	0.38	0.65	0.80	1.05	1.00
	150 tons	0.32	0.01	0.11	0.36	0.59	0.93	0.95
3 dry years	No	1.21	0.37	1.34	1.36	1.35	1.55	1.04
	50 tons	0.91	0.17	0.90	1.06	1.14	1.42	0.99
	100 tons	0.63	0.06	0.48	0.76	0.93	1.30	0.94
	150 tons	0.40	0.04	0.19	0.47	0.72	1.18	0.89

Source: Minte-Vera 2003

Bayesian risk assessments are a tool that can be used to add value to modeling because:

1. They are simple to explain, they provide probability distributions that are easily understandable by managers and users.

2. Their flexibility and generality can be used to deal with very complex problems and support different models.

3. They use prior knowledge and information in a transparent way: it becomes possible to restrict the values of the parameters of the model to biologically plausible values and to take account of information from other systems, therefore it is also good for data poor situations.

4. They account explicitly for uncertainty:

    a. Uncertainty from important but unknown parameters is automatically included

    b. Uncertainty in model choice can be formally incorporated into analysis results by combining the results from different plausible models

    c. Natural variability in the dynamics can easily be incorporated into the models

5. The results can be used simply to project the population into the future under several management scenarios and calculate risk.

6. They facilitate the integration of different approaches such as holistic models and population dynamics models through the use of probabilities as a common 'currency'.

Also, the method forces clear thinking on the management problem since the objectives of management need to be stated clearly and translated into performance measures. Once the managers make a decision, the consequences should be monitored and new data points could be used to update the analysis in order to decrease uncertainty.

Bayesian techniques are not widely used at present, mainly because they are not taught regularly in statistical, modeling or management courses. They are also extremely computationally intensive to apply to complex models and the process of specifying prior distributions can be very time consuming. Clearly, the advantages of using Bayesian methods overcome these disadvantages.

Conclusions

There is a definite need for models to predict the impacts of changes to natural, or established modified, flow regimes on river ecosystems and fisheries. Much of the earlier work on environmental flows has aimed at establishing minimum flow requirements for the survival of the fish stock. In most of the cases that are now arising, however, this is not thought to be helpful and that rather, a continuum of response to changing hydrology is needed by planners and managers to set flows at levels that are optimal among all uses (e.g. Arthington et al. 2006). The environmental flow methodologies and fisheries models assessed here are, therefore, oriented more towards a series of scenario-based predictions rather than the production of a single figure such as a minimum flow.

Because most fisheries in larger rivers exist within a framework of rural livelihoods that use the plant and animal resources of the river floodplain system in a number of ways, including agriculture, any generalized decision-support

framework must have the capacity to trace the interactions of these various uses with the fisheries component (Lorenzen et al. 2007). At the same time more specialized models can explore in more detail the reactions of the fish and fisheries to such changes. The review presented above outlines models and knowledge management systems that permit such predictions to be made.

A great range of models are available that relate flow to population, species or assemblage responses in fish communities. Many of these do very similar things to those discussed above but none were identified that presented clear advantages over the various systems presented in this review. It is preferable, therefore, to work with the tools to hand and not complicate or introduce delays by search for a 'perfect' system. Rather, additional models can be integrated into the overall framework to integrate other models and methods as opportunities arise.

Examination of the strengths and deficiencies of the most comprehensive group of environmental flow methodologies available (i.e., holistic methodologies) suggests DRIFT as currently the most amenable to further modification for consideration of flow requirements for complex river fisheries. DRIFT provides a powerful integrating platform covering a range of outputs including direction of change of all ecosystem components including individual fish species and fisheries, and remains at the core of any assessment for a particular river basin. The DRIFT database and engine can be improved by incorporation of conceptual models, empirical relationships between flow and ecological response (e.g., applying the river classification and benchmarking methods recently outlined in Arthington et al. 2006), and certain Bayesian protocols. The idea is to develop a single integrating model with 'tune-outable' components, models and parameters so specialists can select

FIGURE 18.
Proposed integration of DRIFT, Bayesian networks and age-structured models.

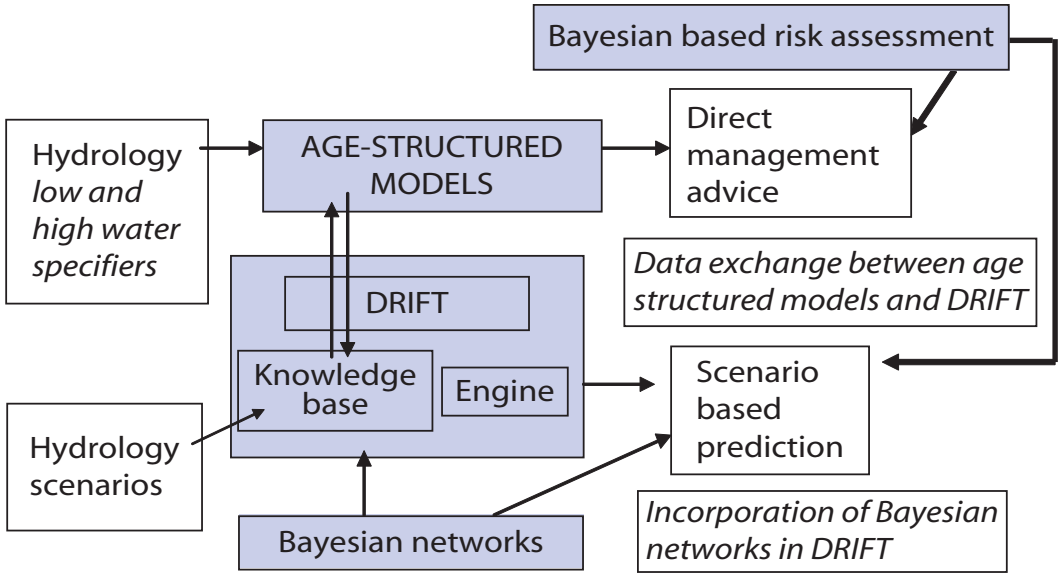


TABLE 11.
Characteristics of the various types of model considered in this report.

Feature	DRIFT	BAYFISH	AGE-STRUCTURED MODELS
Nature	Empirical – can incorporate analytical models	Empirical tool	Analytical tool
Focus	Comprehensive	Comprehensive	Specific
	Links social components	Addresses land use	Incorporates fisheries information
Geographic scale	Site specific, needs to be extrapolated to basin	Basin-wide	Floodplains
Application	Applied 5-6 studies in southern Africa. No floodplains yet	Applied to Mekong upper/ delta and Tonle Sap	Applied 5-6 studies, mainly in Bangladesh
Testing	Implemented but not tested	Not tested, needs to be adapted before testing can be done	Sensitivity analyses done
Hydrological data requirements	Developed using daily hydrological data – but can be adapted to other timescales	Uses discrete hydrological data	Developed using weekly hydrological data – but can be adapted to other timescales
Nature of output	Discrete consequences, but produces a modified time-series for implementation of each scenario	Discrete	Continuous (time series)
	Data comprise outputs of other models, published information, expert opinion	Data comprise expert opinion	Data comprise published density-dependent population processes for fish
Complexity	Complex	Complexity increases rapidly	Simple
Computer	MS Excel based	Programmed	Mainly MS Excel based

the appropriate components, models and parameters depending on where they are situated in a river system, the ecological and fisheries issues under analysis and on the available data. DRIFT is not only an integrating platform but also a communication/training/educational tool and decision-support system to assist in implementation and future management.

The characteristics of the various types of model considered in this review are listed in Table 11 and the way in which the various approaches could interact together is shown in Figure 18. Age-structured models can provide specific fisheries and flow advice on floodplain fisheries, either independently or as an input to the fish components of DRIFT. However, there are systems where age-structured models will not work and there is a need for the DRIFT type of approach based on various information inputs and expert opinion.

Bayesian approaches need to be added to age-structured models in order to give some idea of risk and the possibility of error. Such approaches are also needed for the overall assessment of risk associated with all the approaches examined in this paper.

In order to progress the development of this integrated modeling approach to assessing flow requirements for complex river fisheries, it is recommended that an integrated pilot model combining DRIFT, empirical flow-ecological relationships, Bayesian networks, and age-structured models be developed. This should be tested in pilot river systems and the lessons generated used to both refine the model and identify river systems where other approaches may need to be developed.

Literature Cited

Acreman, M. C.; Farquharson, F. A. K.; McCartney, M. P.; Sullivan, C.; Campbell, K.; Hodgson, N.; Morton, J.; Smith, D.; Birley, M.; Knott, D.; Lazenby, J.; Wingfield, R.; Barbier, E. B. 2000. Managed flood releases from reservoirs: issues and guidance. Report to DFID and the World Commission on Dams. Centre for Ecology and Hydrology, Wallingford, UK. 86 pp.

Allan, J. D.; Abell, R.; Hogan, Z.; Revenga, C.; Taylor, B.; Welcomme, R. L.; Winemiller, K. 2005. Overfishing in inland water. *BioScience* 55: 1041-1051.

Arthington, A. H. 1998. Comparative evaluation of environmental flow assessment techniques: review of holistic methodologies. Land and Water Resources Research and Development Corporation Occasional Paper No. 26/98. Canberra, Australia. 46 pp.

Arthington, A. H.; Balcombe, S. R.; Wilson, G. A.; Thoms, M. C.; Marshall, J. 2005. Spatial and temporal variation in fish assemblage structure in isolated waterholes during the 2001 dry season of an arid-zone river, Cooper Creek, Australia. *Marine and Freshwater Research* 56: 25-35.

Arthington, A. H.; Brizga, S. O.; Choy, S. C.; Kennard, M. J.; Mackay, S. J.; McCosker, R. O.; Ruffini, J. L.; Zalucki, J. M. 2000. *Environmental flow requirements of the Brisbane River downstream of Wivenhoe Dam*. South East Queensland Water Corporation, Brisbane, and Centre for Catchment and In-Stream Research, Griffith University, Brisbane, Australia. 539 pp.

Arthington, A. H.; Brizga, S. O.; Kennard, M. J. 1998. Comparative evaluation of environmental flow assessment techniques: best practice framework. Land and Water Resources Research and Development Corporation Occasional Paper No. 25/98. Canberra, Australia. 26 pp.

Arthington, A. H.; Bunn, S. E.; Poff, N. L.; Naiman, R. J. 2006. The challenge of providing environmental flow rules to sustain river ecosystems. *Ecological Applications* 16: 1311-1318.

Arthington, A. H.; King, J. M.; O'Keeffe, J. H.; Bunn, S. E.; Day, J. A.; Pusey, B. J.; Bluhdorn, D. R.; Tharme, R. 1992. Development of an holistic approach for assessing environmental flow requirements of riverine ecosystems. In: *Proceedings of an International Seminar and Workshop on Water Allocation for the Environment*, eds. J. J. Pigram; B. P. Hooper. Armidale, Australia: Centre for Water Policy Research, University of New England. p. 69-76.

Arthington, A. H.; Lorenzen, K.; Pusey, B. J.; Abell, R.; Halls, A.; Winemiller, K. O.; Arrington, D. A.; Baran, E. 2004a. River fisheries: ecological basis for management and conservation. In: *Proceedings of the Second International Symposium on the Management of Large Rivers for Fisheries Volume I*, eds. R. Welcomme; T. Petr. FAO Regional Office for Asia and the Pacific, Bangkok, Thailand. RAP Publication 2004/16, pp. 21-60.

Arthington, A. H.; Pusey, B. J. 1993. In-stream flow management in Australia: methods, deficiencies and future directions. *Australian Biology* 6: 52-60.

Arthington, A. H.; Pusey, B. J. 2003. Flow restoration and protection in Australian rivers. *River Research and Applications* 19(5-9): 377-395.

Arthington, A. H.; Tharme, R.; Brizga, S. O.; Pusey, B. J.; Kennard, M. J. 2004b. Environmental flow assessment with emphasis on holistic methodologies. In: *Proceedings of the Second International Symposium on the Management of Large Rivers for Fisheries Volume II*, eds. R. Welcomme; T. Petr. FAO Regional Office for Asia and the Pacific, Bangkok, Thailand. RAP Publication 2004/17, pp. 37-65.

Arthington, A. H.; Rall, J. L.; Kennard, M. J.; Pusey, B. J. 2003. Environmental flow requirements of fish in Lesotho Rivers using the DRIFT methodology. *River Research and Applications* 19(5-6): 641-666.

Arthington, A. H.; Zalucki, J. M. eds. 1998. Comparative evaluation of environmental flow assessment techniques: review of methods. Land and Water Resources Research and Development Corporation Occasional Paper No. 27/98. Canberra, Australia. 141 pp.

Balcombe, S.; Arthington, A. H.; Foster, N.; Thoms, M.; Wilson, G.; Bunn, S. E. 2006. Fish assemblages of an Australian dryland river: abundance, assemblage structure and recruitment patterns in the Warrego River, Murray-Darling Basin. *Marine and Freshwater Research* 57: 619-633.

Baran, E.; Cain, J. 2001. Ecological approaches of flood-fish relationships modelling in the Mekong River. Proceedings of the National workshop on Ecological and Environmental Modelling, September 3-4, 2001, Universiti Sains Malaysia, Penang, Malaysia.

Baran, E.; Makin, I.; Baird, I. G. 2003. BayFish: a model of environmental factors driving fish production in the Lower Mekong Basin. Contribution to the Second International Symposium on the Management of Large Rivers for Fisheries, February 11-14, Phnom Penh, Cambodia.

Baron, J. S.; Poff, N. L.; Angermeier, P. L.; Dahm, C. N.; Gleick, P. H.; Hairston, N. G.; Jackson, R. B., Jr.; Johnston, C. A.; Richter, B. D.; Steinman, A. D. 2002. Meeting ecological and societal needs for freshwater. *Ecological Applications* 12: 1247-1260.

Batchelor, C.; Cain, J. 1999. Application of belief networks to water management studies. *Agricultural Water Management* 40(1): 51-57.

Bayes, T. 1763. An essay towards solving a problem in the doctrine of chances. *Philosophical Transactions of the Royal Society of London* 53: 370-418. (reprinted in *Biometrika*. 1958; 45: 293-315).

Bayley, P. B. 1988. Accounting for effort when comparing tropical fisheries in lakes, river-floodplains, and lagoons. *Limnology and Oceanography* 33(4, part 2): 963-972.

Borsuk, M. E.; Burkhardt-Holm, P.; Reichert, P. 2002a. A Bayesian network for investigating the decline in fish catch in Switzerland. iEMSs (International Environmental Modelling and Software Society) Conference proceedings, Lugano, Switzerland.

Borsuk, M. E.; Clemen, R. T.; Maguire, L. A.; Reckhow, K. H. 2001. Stakeholder values and scientific modeling in the Neuse River watershed. *Group Decision and Negotiation* 10: 355-373.

Borsuk, M. E.; Stow, C. A.; Reckhow, K. H. 2002b. Integrative environmental prediction using Bayesian networks: a synthesis of models describing estuarine eutrophication. iEMSs Conference proceedings, Lugano, Switzerland.

Bousquet, F. 1994a. Des milieux, des poissons, des hommes: étude par simulation multi-agent. Le cas de la pêche dans le delta central du Niger. TDM n° 119, Orstom Editions, Paris. 175 pp. http://www.bondy.ird.fr/pleins_textes/index.htm

Bousquet, F. 1994b. Distributed artificial intelligence and object-oriented modelling of a fishery. *Mathematical Computer Modelling* 2018: 97-107.

Bousquet, F.; Barreteau, O.; Le Page, C.; Mullon, C.; Weber, J. 1999. An environmental modelling approach: the use of multi-agent simulations. In: *Advances in environmental and ecological modelling*, eds. F. Blasco; A. Weill. Elsevier.

Bousquet, F.; Cambier, C.; Mullon, C.; Morand, P.; Quensière, J.; Pave, A. 1993. Simulating the interaction between a society and a renewable resource. *Journal of Biological Systems* 1(2): 199-214. http://www.bondy.ird.fr/pleins_textes/index.htm

Bovee, K. D. 1982. A guide to stream habitat analysis using the instream flow incremental methodology. Instream Flow Information Paper 21. FWS/OBS-82/26. U. S. Department of Fisheries and Wildlife Services, Washington DC; 248 pp.

Brizga, S. O. 2000. Burnett Basin Water Allocation and Management Plan: Proposed Environmental Flow Performance Measures. Brisbane: Department of Natural Resources, 2 volumes.

Brizga, S. O.; Arthington, A. H.; Pusey, B. J.; Kennard, M. J.; Mackay, S. J.; Werren G. L.; Craigie, N. M.; Choy, S. J. 2002. Benchmarking, a 'top-down' methodology for assessing environmental flows in Australian rivers. ENVIRO FLOWS 2002. Proceedings of the International Conference on Environmental Flows for River Systems, incorporating the 4th International Ecohydraulics Symposium. Unpublished proceedings. Cape Town, March 2002.

Brown, C. A.; Joubert, A. 2003. Using multicriteria analysis to develop environmental flow scenarios for rivers targeted for water resource development. *Water SA* 29(4): 365-374.

Brown, C. A.; King, J. M. 2002. Breede River Basin Study. DRIFT Application. Unpublished Southern Waters Report for Department of Water Affairs and Forestry, and Water Research Commission. 77 pp. Available at http://www.southernwaters.co.za.

Brown, C. A.; King, J. M.; Sabet, H. 2000a. Project Structure and Methods. Lesotho Highlands Water Project. Contract LHDA 648: Consulting services for the establishment and monitoring of instream flow requirements for river courses downstream of LHWP dams. Report No. LHDA 648-F-03. *c.* 200 pp.

Brown, C.; Pemberton, C.; Birkhead, A.; Bok, A.; Boucher, C.; Dollar, E.; Harding, W.; Kamish, W.; King, J.; Paxton, B.; Ractliffe, S. 2006. In Support of Water-resource planning – highlighting key management issues using DRIFT: A Case study. *Water SA* 32(2): 181-191.

Brown, C.; Pemberton, C.; Greyling, A.; King, J. M. 2005. DRIFT USER MANUAL: VOLUME 1: Biophysical Module for predicting overall river condition in small to medium sized rivers with predictable flow regimes. *WRC Final Report*, May 2005.

Brown, C. A.; Sparks, A.; Howard, G. 2000b. Palmiet River Instream Flow Assessment: IFR for the Riverine Ecosystem Proceedings of the IFR Workshop and determination of associated dam yields. Report to Department of Water Affairs and Forestry. No. G400-00-0499. 168 pp.

Bunn, S. E. 1998. Recent approaches to assessing and providing environmental flows: concluding comments. Pp. 123-129 in *Water for the environment: Recent approaches to assessing and providing environmental flows*. Proceedings of AWWA Forum, eds. A. H. Arthington; J. M. Zalucki. Australian Water and Wastewater Association (AWWA), Brisbane, Australia.

Bunn, S. E.; Arthington, A. H. 2002. Basic principles and consequences of altered hydrological regimes for aquatic biodiversity. *Environmental Management* 30: 492-507.

Burgess, G. K.; Vanderbyl, T. L. 1996. Habitat analysis method for determining environmental flow requirements. Pp. 83-92. In: *Water and the environment*. Proceedings 23rd hydrology and water resources symposium, Hobart. Institution of Engineers, Barton, Australian Capital Territory.

Cain, J. 2001. Planning improvements in natural resources management. Guidelines for using Bayesian networks to support the planning and management of development programmes in the water sector and beyond. Centre for Ecology & Hydrology (CEH) Wallingford, UK. 124 pp.

Cambier, C.; Bousquet, F.; Dansoko, D. 1993. Un univers multi-agents pour la modelisation du systeme de la peche du delta central du Niger. In: *Actes du 1er colloque africain sur la recherche en informatique,* ed. M. Tchuente. *INRIA, Rocquencourt* 111-122. http://www.bondy.ird.fr/pleins_textes/index.htm

Charniak, E. 1991. Bayesian networks without tears. *Artificial Intelligence* 12(4): 50-63. http://www-psych.stanford.edu/~jbt/224/Charniak_91.pdf

Chen, Y.; Breen, P. A.; Andrew, N. L. 2000. Impacts of outliers and mis-specification of priors on Bayesian fisheries stock assessment. *Canadian Journal of Fisheries and Aquatic Sciences* 57: 2293-2305.

Christensen, V.; Pauly, D. 1992. Ecopath II - a system for balancing steady-state ecosystem models and calculating network characteristics. *Ecological Modelling* 61: 169-185.

Christensen, V.; Pauly, D. eds. 1993. Trophic models of aquatic ecosystems. *ICLARM Conference Proceedings* 26. 390 pp.

Cottingham, P.; Thoms, M. C.; Quinn, G. P. 2002. Scientific panels and their use in environmental flow assessment in Australia. *Australian Journal of Water Resources* 5: 103-111.

Davies, P. M.; Bunn, S. E.; Arthington, A. H. 1996. Environmental water requirements for lowland river systems on the Swan Coastal Plain. Unpublished report to the Water and Rivers Commission, Western Australia. Department of Zoology, University of Western Australia: Perth; 114 pp.

DWAF (Department of Water Affairs and Forestry). 1999. *Resource directed measures for protection of water resources*. *Volume 2: integrated manual. Version 1.0.* Institute for Water Quality Studies, Department of Water Affairs and Forestry, Pretoria, South Africa.

Doran, J. E. 1997. Distributed Artificial Intelligence and emergent social complexity. In: *Time, process and structured transformation in archaeology*, eds. S. E. van der Leeuw; J. McGlade. London & New York: Routledge (One world archaeology series, Vol. 26). pp. 283-297.

Downes, B. J.; Barmuta, L. A.; Fairweather, P. G.; Faith, D. P.; Keough, M. J.; Lake, P. S.; Mapstone, B. D.; Quinn, G. P. 2002. *Monitoring ecological impacts: Concepts and practice in flowing waters.* Cambridge University Press, Cambridge, UK.

Dudgeon, D.; Arthington, A. H.; Gessner, M. O.; Kawabata, Z.; Knowler, D.; Lévêque, C.; Naiman, R. J.; Prieur-Richard, A-H.; Soto, D.; Stiassny, M. L. J.; Sullivan, C. A. 2006. Freshwater biodiversity: importance, status, and conservation challenges. *Biological Reviews* 81: 163-182.

Dunbar, M. J.; Gustard, A.; Acreman, M. C.; Elliott, C. R. N. 1998. Review of overseas approaches to setting river flow objectives. *Environment Agency R&D Technical Report* W6B(96)4. Institute of Hydrology, Wallingford, United Kingdom. 80 pp.

Ellison, A. M. 1996. An introduction to Bayesian inference for ecological research and environmental decision-making. *Ecological Applications* 6(4): 1036-1046.

Ferber, J. 1999. Multi-agent systems: an introduction to Distributed Artificial Intelligence. Reading, MA: Addison-Wesley.

Fox, W. W. 1970. An exponential surplus-yield model for optimizing exploited fish populations. *Transactions of the American Fisheries Society* 99: 80-88.

Gippel, C. J.; Stewardson, M. J. 1998. Use of wetted perimeter in defining minimum environmental flows. *Regulated Rivers: Research & Management* 14: 53-67.

Growns, I. O.; Kotlash, A. 1994. Environmental flow allocations for the Hawkesbury-Nepean River system: a review of information. *Australian Water Technologies EnSight Report No*. 94/189. 55 pp.

Halls, A. S.; Debnath, K.; Kirkwood, G. P.; Payne, I. A. 2000. Density-dependent recruitment of *Puntius sophore*, in floodplain waterbodies in Bangladesh. *Journal of Fish Biology* 56: 905-914.

Halls, A. S.; Hoggarth, D. D.; Debnath, K. 1998. Impact of flood control schemes on river fish migrations and species assemblages in Bangladesh. *Journal of Fish Biology* 53 (Suppl. A): 358-380.

Halls, A. S.; Hoggarth, D. D.; Debnath, K. 1999. Impacts of hydraulic engineering on the dynamics and production potential of floodplain fish populations in Bangladesh. *Fisheries Management and Ecology* 6: 261-285.

Halls, A. S.; Kirkwood, G. P.; Payne, A. I. 2001. A dynamic pool model for floodplain-river fisheries. *Ecohydrology and Hydrobiology* 1(3): 323-339.

Halls, A. S.; Welcomme, R. L. 2004. Dynamics of river fish populations in response to hydrological conditions: A simulation study. *River Research and Applications* 20: 985-1000.

Halls, A. S.; Welcomme, R. L.; Burn, R. W. 2006. The relationship between multispecies catch and effort: Among fishery comparisons. *Fisheries Research* 77: 78-83.

Hilborn, R.; Walters, C. J. 1992. Quantitative Fisheries Stock Assessment. Chapman Hall, London. 570pp.

Hilborn, R.; Mangel, M. 1997. The Ecological Detective. Princeton, Princeton University Press.

Hobbs, B. F. 1997. Bayesian methods for analysing climate change and water resource uncertainties. *Journal of Environmental Management* 49: 53-72.

Holling, C. S. 2001. Understanding the complexity of economic, ecological, and social systems. *Ecosystems* 4: 390-405.

Janssen, M. A. 2002. Modelling human dimensions of global environmental change. Pp. 394–408 in *Social and economic dimensions of global environmental change*, ed. P. Timmerman. Encyclopedia of Global Environmental Change Volume 5. John Wiley & Sons.

Jensen, F. V. 1996. An introduction to Bayesian networks. UCL press, London. 178 pp.

Jordan, M. I. 1999. Learning in graphical models. MIT Press, Cambridge.

Jowett, I. G. 1997. Instream flow methods: a comparison of approaches. *Regulated Rivers: Research & Management* 13: 115-127.

Junk, W. J.; Bayley, P. B.; Sparks, R. E. 1989. The flood pulse concept in river-floodplain systems. Proceedings of the International Large River Symposium (LARS). *Canadian Special Publications in Fisheries and Aquatic Sciences* 106: 110-127.

Kennard, M. J. 2000. Freshwater Fish. In: *Burnett Basin WAMP: Current Environmental Conditions and Impacts of Existing Water Resources Development. Volume II(b), Appendix G*, eds. S. Brizga; A. Arthington; S. Choy; L. Duivenvoorden; M. Kennard; R. Maynard; W. Poplawski. Queensland Government Natural Resources and Mines, Brisbane. 78 pp.

Kennard, M. J.; Harch, B. D.; Pusey, B. J.; Arthington, A. H. 2006a. Accurately defining the reference condition for summary biotic metrics: a comparison of four approaches. *Hydrobiologia* 572: 151-170.

Kennard, M. J.; Pusey, B. J.; Arthington, A. H.; Harch, B. D.; Mackay, S. J. 2006b. Development and application of a predictive model of freshwater fish assemblage composition to evaluate river health in eastern Australia. *Hydrobiologia* 572: 33-57.

King, J. M.; Brown, C. A.; Sabet, H. 2003. A scenario-based holistic approach to environmental flow assessments for regulated rivers. *River Research and Applications* 19: 619-640.

King, J. M.; Louw, M. D. 1998. Instream flow assessments for regulated rivers in South Africa using the Building Block Methodology. *Aquatic Ecosystem Health and Management* 1: 109-124.

King, J. M; Tharme, R. E. 1994. Assessment of the instream flow incremental methodology and initial development of alternative instream flow methodologies for South Africa. *Water Research Commission Report No.* 295/1/94. Water Research Commission, Pretoria. 590 pp.

King, J. M.; Tharme, R. E.; Brown, C. A. 1999. Definition and implementation of instream flows. Thematic Report for the World Commission on Dams. Southern Waters Ecological Research and Consulting, Cape Town, South Africa. 63 pp.

King, J. M.; Tharme, R. E.; De Villiers, M. eds. 2000a. Environmental flow assessments for rivers: manual for the Building Block Methodology. *Water Research Commission Technology Transfer Report No.* TT131/00. Water Research Commission, Pretoria. 340 pp.

King, J. M.; Sabet, H.; Brown, C. A.; Hirst, S. 2000b. Final Report: Summary of Main Findings. Lesotho Highlands Water Project. Contract LHDA 648: Consulting services for the establishment and monitoring of instream flow requirements for river courses downstream of LHWP dams. Report No. LHDA 648-F-02. 75 pp.

Kuikka, S.; Hildén, M.; Gislason, H.; Hansson, S.; Sparholt, H.; Varis, O. 1999. Modeling environmentally driven uncertainties in Baltic cod (*Gadus morhua*) management by Bayesian influence diagrams. *Canadian Journal of Fisheries and Aquatic Sciences* 56(4): 629-641.

Lae, R. 1992a. Impact des barrages sur les pêcheries artisanales du Delta Centrale du Niger. *Cahiers d'agricultures* 2: 14-21.

Lae, R. 1992b. Influence de l'hydrobiologie sur l'évolution des pêcheries du Delta Centrale du Niger de 1966 a 1989. *Aquatic Living Resources* 52: 115-126.

Lee, D. C.; Rieman, B. E. 1997. Population viability assessment of salmonids by using probabilistic networks. *North American Journal of Fisheries Management* 17: 1144-1157.

LHDA (Lesotho Highlands Development Authority). 2003. Procedures for the implementation of the LHWP (Lesotho Highlands Water Project) Phase 1. Instream Flow Requirements Policy. 50 pp.

Loneragan, N. R.; Bunn, S. E. 1999. River flows and estuarine ecosystems: implications for coastal fisheries from a review and a case study of the Logan River, southeast Queensland. *Australian Journal of Ecology* 24(4): 431-440.

Lorenzen, K.; Enberg, K. 2002. Density-dependent growth as a key mechanism in the regulation of fish populations: evidence from among-population comparisons. *Proceedings of the Royal Society of London Biological Sciences* 269: 49-54.

Lorenzen, K. D.; De Graaf, G.; Halls, A. S. 2002. *A biomass dynamics model for assessing interactions between fisheries and water management in highly seasonal water bodies*. Report. London: Imperial College, London.

Lorenzen, K.; Smith, L.; Nguyen Khoa, S.; Burton, M.; Garaway, C. 2007. *Guidance Manual: Management of Irrigation Development Impacts on Fisheries*. International Water Management Institute / WorldFish Center. 259 pp.

Marcot, B. G.; Holthausen, R. S.; Raphael, M. G.; Rowland, M.; Wisdom, M. 2001. Using Bayesian belief networks to evaluate fish and wildlife population viability under land management alternatives from an environmental impact statement. Forest Ecology and Management 153(1-3): 29-42. http://www.onid.orst.edu/~gordons/docs/Marcot-2001.pdf

McAllister, M. K.; Pikitch, E. K.; Punt, A. E.; Hilborn, R. 1994. A Bayesian approach to stock assessment and harvest decisions using the sampling/importance re-sampling algorithm. *Canadian Journal of Fisheries and Aquatic Sciences* 51: 2673-2687.

Metsi Consultants. 2000a. Consulting services for the establishment and monitoring of the instream flow requirements for river courses downstream of LHWP dams. IFR Methodology Report No. 648-F-03 (Volume 2). Authors: C. Brown; J. King. Lesotho Highlands Development Authority Contract 648, Maseru, Lesotho.

Metsi Consultants. 2000b. Consulting services for the establishment and monitoring of the instream flow requirements for river courses downstream of LHWP dams. Fish. Report No. 648-F-18. Authors: A. Arthington; J. Rall; M. Kennard. Lesotho Highlands Development Authority Contract 648: Maseru, Lesotho; 85 pp (plus Appendices).

Meyer, R.; Millar, R. B. 1999. Bayesian stock assessment using a state-space implementation of the delay difference model. *Canadian Journal of Fisheries and Aquatic Sciences* 56: 37-52.

Milhous, R. T.; Updike, M. A.; Schneider, D. M. 1989. Physical habitat simulation system reference manual - version 2. *Instream Flow Information Paper No.* 26. *U. S. D. I. Fish and Wildlife Service Biological Report* 89(16).

Minte-Vera, C. V. 2003. Risk assessment of a floodplain migratory fish subject to fisheries and dam-operation procedures. Contribution to the Second International Symposium on the Management of Large Rivers for Fisheries, February 11-14, Phnom Penh, Cambodia.

Morand, P.; Bousquet, F. 1994. Relations entre l'effort de pêche, la dynamique du peuplement ichthyologique et le niveau des captures dans un système fleuve-plaine. Pp. 267-281 in La Pêche dans le Delta Central du Niger, ed. J. Quensière. Paris, ORSTOM.

Moreau, J. 1980. Essai d'application au lac Alaotra (Madagascar) d'un modèle d'étude de pêcheries pour les plaines d'inondation intertropicales. *Cah ORSTOM* (*Hydrobiol.*), 13(1-2): 83-91.

Moss, S.; Downing, T. E.; Rouchier, J. 2000. Demonstrating the role of stakeholder participation: an agent based social simulation model of water demand policy response. Report No 76. Centre for Policy Modelling, Manchester Metropolitan University Business School.

Muller, W. J. ed. 1997. Vaal River system analysis update: environmental flow management plan for the Vaal River. Unpublished report prepared for the Department of Water Affairs and Forestry and BKS Incorporated.

Muncy, R. J. 1978. An evaluation of the Zambian Kafue River floodplain fishery. *CIFA Technical Paper*. 5: 153-164.

Naiman, R. J.; Bunn, S. E.; Nilsson, C.; Petts, G. E.; Pinay, G.; Thompson, L. C. 2002. Legitimizing fluvial ecosystems as users of water. *Environmental Management* 30: 455-467.

O'Keeffe, J.; Hughes, D.; Tharme, R. E. 2002. Linking ecological responses to altered flows, for use in environmental flow assessments: the Flow Stressor-Response method. *Verhandhingen Internationale Vereinigung Limnologie* 28: 84-92.

Olden, J. D.; Poff, N. L. 2003. Redundancy and the choice of hydrological indices for characterising streamflow regimes. *River Research and Applications* 19(2): 101-121.

Ostrom, E.; Gardner, R.; Walker, J. 1994. Rules, Games, and Common-Pool Resources. The University of Michigan Press.

Peterson, J. T.; Evans, J. W. 2003. Quantitative decision analysis for sport fisheries management. *Fisheries* 28(1): 10-21.

Pettit, N. E.; Froend, R. H.; Davies, P. M. 2001. Identifying the natural flow regime and the relationship with riparian vegetation for two contrasting Western Australian rivers. *Regulated Rivers: Research and Management* 17: 201-215.

Petts, G. E.; Bickerton, M. A.; Crawford, C.; Lerner, D. N.; Evans, D. 1999. Flow management to sustain groundwater-dominated stream ecosystems. *Hydrological Processes* 13: 497-513.

Poff, N. L.; Allan, J. D.; Bain, M. B.; Karr, J. R.; Prestegaard, K. L.; Richter, B. D.; Sparks, R. E.; Stromberg, J. C. 1997. The natural flow regime. A paradigm for river conservation and restoration. *BioScience* 47: 769-784.

Poff, N. L.; Allan, J. D.; Palmer, M. A.; Hart, D. D.; Richter, B. D.; Arthington, A. H.; Rogers, K. H.; Meyer, J. L.; Stanford, J. A. 2003. River Flows and Water Wars: Emerging Science for Environmental Decision-making. *Frontiers in Ecology and the Environment* 1(6): 298-306.

Postel, S.; Richter, B. 2003. Rivers for Life: Managing Water for People and Nature. Island Press, Washington, DC.

Punt, A. E.; Hilborn, R. 1997. Fisheries stock assessment and decision analysis: the Bayesian approach. *Reviews in Fish Biology and Fisheries* 7: 35-63.

Punt, A. E.; Hilborn, R. 2001. BAYES-SA - Bayesian stock assessment methods in fisheries - User's manual. *FAO Computerized Information Series (Fisheries) No.* 12. Rome: FAO. 56 pp.

Pusey, B. J. 1998. Methods addressing the flow requirements of fish. In: *Comparative evaluation of environmental flow assessment techniques: review of methods*, eds. A. H. Arthington; J. M. Zalucki. Land and Water Resources Research and Development Corporation Occasional Paper No. 27/98. Canberra, Australia. p. 66-105.

Pusey, B. J.; Arthington, A. H.; Kennard, M. J. 2004. Hydrologic regime and its influence on broad-scale patterns of fish biodiversity in north-eastern Australian rivers. *Proceedings of the Fifth International Symposium on Ecohydraulics. Aquatic Habitats, Analysis and Restoration*. Madrid, Spain, pp. 75-81.

Pusey, B. J.; Arthington, A. H.; Read, M. G. 1995. Species richness and spatial variation in fish assemblage structure in two rivers of the Wet Tropics of Northern Queensland, Australia. *Environmental Biology of Fishes* 42: 181-199.

Pusey, B. J.; Arthington, A. H.; Read, M. G. 1998. Freshwater fishes of the Burdekin River, Australia: biogeography, history and spatial variation in assemblage structure. *Environmental Biology of Fishes* 53: 303-318.

Pusey, B. J.; Kennard, M. J.; Arthington, A. H. 2000. Discharge variability and the development of predictive models relating stream fish assemblage structure to habitat in north-eastern Australia. *Ecology of Freshwater Fishes* 9: 30-50.

Reckhow, K. H. 2002. Bayesian approaches in ecological analysis and modeling. In: *The role of models in ecosystem science*, eds. C. D. Canham; J. J. Cole; W. K. Lauenroth. Princeton University Press.

Revenga, C.; Brunner, J.; Henninger, N.; Kassem, K.; Payne, R. 2000. *Pilot analysis of global ecosystems: freshwater ecosystems*. World Resources Institute, Washington, DC. 83 pp.

Richter, B. D.; Baumgartner, J. V.; Powell, J.; Braun, D. P. 1996. A method for assessing hydrological alteration within ecosystems. *Conservation Biology* 10(4): 1163-1174.

Richter, B. D.; Baumgartner, J. V.; Wigington, R.; Braun, D. P. 1997. How much water does a river need? *Freshwater Biology* 37: 231-249.

Rieman, B.; Peterson, J. T.; Clayton, J.; Howell, P.; Thurow, R.; Thompson, W.; Lee, D. 2001. Evaluation of potential effects of federal land management alternatives on trends of salmonids and their habitats in the interior Columbia River basin. *Forest Ecology and Management* 153: 43-62. http://www.srs.fs.usda.gov/pubs/viewpub.jsp?index=3657

Rosenberg, D. M.; Mccully, P.; Pringle, C. M. 2000. Global-scale environmental effects of hydrological alterations: introduction. *BioScience* 50(9): 746-751.

Sabet, H.; Brown, C.; Hirst, S. 2002. Additional scenarios and production of a new final report to augment consulting services for the establishment and monitoring of the instream flow requirements for river courses downstream of LHWP dams. Final Report. Summary of Main Findings for Phase 1 Development. LHDA Report No. 678-F-001.

Schumm, S. A. 1969. River metamorphosis. Journal of the Hydraulics Division. *Proceedings of the American Society of Civil Engineers* 95: 251-273.

Shankar, B.; Halls, A.; Barr, J.; Rahman, M. 2002. Management strategies to improve floodplain livelihoods in Bangladesh: some modelling results. Proceedings of the Wetlands Conference, IUCN, Bangladesh, July 2002.

Shankar, B.; Halls, A. S.; Barr, J. 2003. Fish out of water: modelling tradeoffs between agriculture and fisheries in the floodplains of Bangladesh. Contribution to the Second International Symposium on the Management of Large Rivers for Fisheries, February 11-14, Phnom Penh, Cambodia.

Shepard, B. B.; Sanborn, B.; Ulmer, L.; Lee, D. C. 1997. Status and risk of extinction for westslope cutthroat trout in the Upper Missouri River Basin, Montana. *North American Journal of Fisheries Management* 17: 1158-1172.

SKM (Sinclair Knight Merz Consulting). 2001. Determination of the Environmental Water Requirements of the Onkaparinga River Catchment. Proposed Approach for Stages 3 and 4 of the Project. Report for Onkaparinga Catchment Water Management Board. Adelaide: SKM Consulting.

Soncini-Sessa, R.; Castelletti, A.; Weber, E. 2002. Participatory decision making in reservoir planning. iEMSs Conference proceedings, Lugano, Switzerland.

Sparks, R. E. 1992. Risks of altering the hydrologic regime of large rivers. In: *Predicting ecosystem risk* Vol XX, eds. J. Cairns; B. R. Niederlehner; D. R. Orvos. Princeton, N.J.: Princeton Scientific Publishing Co. Pp. 119-152.

Sparks, R. E. 1995. Need for ecosystem management of large rivers and floodplains. *BioScience* 45: 168-182.

Stalnaker, C.; Lamb, B. L.; Henriksen, J.; Bovee, K.; Bartholow, J. 1994. The Instream Incremental Methodology. A Primer for IFIM. Biological Report 29. U. S. Department of the Interior, National Biological Service, Washington, DC.

Steward, H. J.; Madamombe, E. K.; Topping, C. C. 2002. Adapting environmental flow methodologies for Zimbabwe. Unpublished Paper. *Environmental Flows for River Systems Working Conference and Fourth International Ecohydraulics Symposium*. March 3-8, 2002, Cape Town, South Africa.

Stewardson, M. 2001. The flow events method for developing environmental flow regimes. In: *The value of healthy rivers*, eds. I. Rutherfurd; F. Sheldon; G. Brierley; C. Kenyon. Proceedings of the Third Australian Stream Management Conference. August 27-29, 2001, Brisbane, Queensland, Australia. Pp. 577-582.

Stewardson, M.; Gippel, C. 1997. In-stream environmental flow design: a review. Draft Report. Cooperative Research Centre for Catchment Hydrology, Department of Civil and Environmental Engineering, University of Melbourne. Victoria, Australia. 99 pp.

Swales, S.; Harris, J. H. 1995. The expert panel assessment method (EPAM): a new tool for determining environmental flows in regulated rivers. Pp. 125-134 in *The ecological basis for river management,* eds. D. M. Harper; A. J. D. Ferguson. John Wiley & Sons, New York. 614 pp.

Tennant, D. L. 1976. Instream flow regimens for fish, wildlife, recreation and related environmental resources. *Fisheries* 1(4): 6-10.

Tharme, R. E. 1996. Review of international methodologies for the quantification of the instream flow requirements of rivers. Water law review final report for policy development. For the Department of Water Affairs and Forestry, Pretoria, South Africa. Freshwater Research Unit, University of Cape Town. 116 pp.

Tharme, R. E. 2003. A global perspective on environmental flow assessment: emerging trends in the development and application of environmental flow methodologies for rivers. *River Research and Applications* 19(5-6): 397-441.

Tharme, R. E.; King, J. M. 1998. Development of the Building Block Methodology for instream flow assessments, and supporting research on the effects of different magnitude flows on riverine ecosystems. *Water Research Commission Report* No. 576/1/98. 452 pp.

Thoms, M. C.; Sheldon, F.; Roberts, J.; Harris, J.; Hillman, T. J. 1996. Scientific panel assessment of environmental flows for the Barwon-Darling River. A report to the Technical Services Division of the New South Wales Department of Land and Water Conservation. 161 pp.

Thorp, J. H.; Delong, M. D. 1994. The riverine productivity model: an heuristic view of carbon sources and organic matter processing in large river ecosystems. *Oikos* 70: 305-308.

Vannote, R. L.; Minshall, G. W.; Cummins, K. W.; Sedell, J. R.; Cushing, C. E. 1980. The River Continuum Concept. *Canadian Journal of Fisheries and Aquatic Sciences* 37: 130-137.

Van Zalinge, N.; Loeung, D.; Pengbun, N.; Sarkkula, J.; Koponen, J. 2003. Mekong flood levels and Tonle Sap fish catches. Contribution to the Second International Symposium on the Management of Large Rivers for Fisheries, February 11-14, Phnom Penh, Cambodia.

Varis, O. 1998. A belief network approach to optimization and parameter estimation: application to resource and environmental management. *Artificial Intelligence* 101(1-2): 135-163.

Varis, O.; Kettunen, J.; Sirvi, H. 1990. Bayesian influence diagrams approach to complex environmental management including observational design. *Computational statistics and data analysis* 9: 77-91.

Varis, O.; Lahtela, V. 2002. Integrated water resources management along the Senegal river: introducing an analytical framework. *Water Resources Development* 18(4): 501-521.

Wade, P. R. 2000. Bayesian methods in conservation biology. *Conservation Biology* 14(5): 1308-1316.

Walter, A. C.; Burgess, G. K.; Johnston, P. J. 1994. Assessment of a process for determining environmental flows. Pp. 195-201 in *Environmental flows seminar proceedings*. AWWA Inc., Artarmon, Australia.

Walters, C. J.; Christensen, V.; Pauly, D. 1997. Structuring dynamic models of exploited ecosystems from trophic mass-balance assessments. *Reviews in Fish Biology and Fisheries* 7: 137-172.

Ward, J. V.; Stanford, J. A. 1983. The serial discontinuity concept of lotic ecosystems. Pp 29-42 in *Dynamics of lotic ecosystems*, eds. T. D. Fontaine III; S. M. Bartell. Ann Arbor Science Publishers, Ann Arbor, Michigan. 494 pp.

Ward, J. V.; Tockner, U.; Uehlinger, U.; Malard, F. 2001. Understanding natural patterns and processes in river corridors as the basis for effective river restoration. *Regulated Rivers Research and Management* 117: 311-323.

Weisbuch, G.; Duchateau-Nguyen, G. 1998. Societies, cultures and fisheries from a modeling perspective. *Journal of Artificial Societies and Social Simulatio*n vol. 1, no. 2. http://www.soc.surrey.ac.uk/JASSS/1/2/2.html

Welcomme, R. L. 1975. The fisheries ecology of African Floodplains, *CIFA Technical Paper* 3: 51 pp.

Welcomme, R. L. 1985. River Fisheries. FAO Fisheries Technical Paper 262. 330 pp.

Welcomme, R. L. 2001. *Inland Fisheries: Ecology and Management*. Fishing News Books, Blackwell Scientific, Oxford, 358 pp.

Welcomme, R. L.; Bene, C.; Brown, C. A.; Arthington, A.; Dugan, P.; King, J. M.; Sugunan, V. 2006. Predicting the water requirements of river fisheries. In: *Wetlands and Natural Resource Management*, eds. J. T. A. Verhoeven; B. Beltman; R. Bobbink; D. F Whigham. Ecological Studies Vol. 190, Springer-Verlag, Berlin, Heidelberg. p. 123-154.

Welcomme, R. L.; Hagborg, D. 1977. Towards a model of a floodplain fish population and its fishery. *Environmental Biology of Fishes* 2(1): 7-24.

Winston, W. L. 1994. Operations Research: Applications and Algorithms. Duxbury Press, California.

4. *Does International Cereal Trade Save Water? The Impact of Virtual Water Trade on Global Water Use.* Charlotte de Fraiture, David Molden, Mark Rosegrant, Ximing Cai and Upali Amarasinghe. 2004.

5. *Evolution of Irrigation in South and Southeast Asia.* Randolph Barker and François Molle. 2004.

6. *Macro Policies and Investment Priorities for Irrigated Agriculture in Vietnam.* Randolph Barker, Claudia Ringler, Nguyen Minh Tien and Mark Rosegrant. 2004.

7. *Impacts of Irrigation on Inland Fisheries: Appraisals in Laos and Sri Lanka.* Sophie Nguyen-Khoa, Laurence Smith and Kai Lorenzen. 2005.

8. *Meta-Analysis to Assess Impact of Watershed Program and People's Participation.* P.K. Joshi, A.K. Jha, S.P. Wani, Laxmi Joshi and R.L. Shiyani. 2005.

9. *Historical Transformations of the Lower Jordan River Basin (in Jordan): Changes in Water Use and Projections (1950–2025).* Rémy Courcier, Jean-Philippe Venot and François Molle. 2005.

10. *Cities versus Agriculture: Revisiting Intersectoral Water Transfers, Potential Gains and Conflicts.* François Molle and Jeremy Berkoff. 2006.

11. *Prospects for Productive Use of Saline Water in West Asia and North Africa.* John Stenhouse and Jacob W. Kijne. 2006.

12. *Impact of Land Use on River Basin Water Balance: A Case Study of the Modder River Basin, South Africa.* Y.E. Woyessa, E. Pretorius, P.S. van Heerden, M. Hensley and L.D. van Rensburg. 2006.

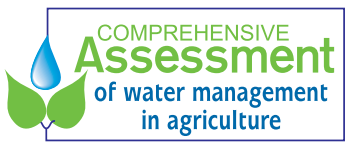
13. *Intensifying Agricultural Sustainability: An Analysis of Impacts and Drivers in the Development of 'Bright Spots'. A.* D. Noble, D. A. Bossio, F. W. T. Penning de Vries, J. Pretty and T. M. Thiyagarajan. 2006.

14. *Multifunctional Agricultural Policy; Reduced Domestic Support; and Liberalized Trade: An Empirical Assessment for Taiwanese Rice.* Richard N. Boisvert and Hung Hao Chang. 2006.

15. *Tropical River Fisheries Valuation: A Global Synthesis and Critical Review.* Arthur E. Neiland, and Christophe Béné. 2006.

16. *Planning and Managing Water Resources at the River-Basin Level: Emergence and Evolution of a Concept.* François Molle. 2006.

17. *Water Requirements of Floodplain Rivers and Fisheries: Existing Decision Support Tools and Pathways for Development.* A. H. Arthington, E. Baran, C. A. Brown, P. Dugan, A. S. Halls, J. M. King, C. V. Minte-Vera, R. E. Tharme and R. L. Welcomme. 2007.



Postal Address: IWMI, P O Box 2075, Colombo, Sri Lanka **Location:** 127 Sunil Mawatha, Pelawatte, Battaramulla, Sri Lanka
Telephone: +94-11 2880000, 2787404 **Fax:** +94-11 2786854
Email: comp.assessment@cgiar.org **Website:** www.iwmi.org/assessment

ISSN 1391-9407
ISBN 978-92-9090-656-8