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Environmental flows in integrated water resources management: Linking flows, services and values

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Ph.D. Thesis

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Preface

This thesis "**Environmental Flows in Integrated Water Resources Management: Linking Flows, Services and Values**" has been submitted as part of the requirements for obtaining the Ph.D. Degree. The study has been carried out from September 2003 to October 2006, at:

- Institute of Environment & Resources, Technical University of Denmark, Kongens Lyngby, Denmark, under the supervision of Prof. Dan Rosbjerg,
- Water Management Department, DHI Water & Environment, Hørsholm, Denmark, under the supervision of Prof. Torkil Jønch-Clausen,
- Department of Policy Analysis, National Environmental Research Institute, Roskilde, Denmark, under the supervision of Dr. Jesper Sølver Schou, and
- International Water Management Institute (IWMI), Colombo, Sri Lanka, under the supervision of Dr. Vladimir Smakhtin

The Ph.D. thesis is accompanied by two peer-reviewed papers published in conference proceedings, two submitted journal papers, and material produced in relation to establishing the Global Environmental Flows Network:

Paper A
Korsgaard, L., Jønch-Clausen, T., Rosbjerg, D. & Schou, J.S. (2005): Quantification of environmental flows in integrated water resources management. In: Brebbia, C.A. & Antunes do Carmo, J.S. (eds.): River Basin Management III, WIT Press, Boston. 141-150.

Paper B
Korsgaard, L., Jønch-Clausen, T., Rosbjerg, D. & Schou, J.S. (in press): Using economic valuation of environmental flows to integrate ecological aspects into water management. Proceedings of the 3rd International Symposium on Integrated Water Resources Management, 26-28 September 2006, Bochum, Germany.

Paper C
Korsgaard, L. & Schou, J.S. (submitted): Economic valuation of aquatic ecosystem services in developing countries. Submitted to *Ecological Economics.*

Paper D
Korsgaard, L., Jensen, R.A., Jønch-Clausen, T., Rosbjerg, D. & Schou, J.S. (submitted): A service and value based approach to estimating Environmental Flows. Submitted to *International Journal of River Basin Management.*

Material produced for the Global Environmental Flows Network
Concept note.
Programme for Seminar at World Water Week in Stockholm.
Newsletter.

The papers and network material are not included in this www-version but may be obtained from the Library at the Institute of Environment & Resources, Bygningstorvet, Building 115, Technical University of Denmark, DK-2800 Kgs. Lyngby (library@er.dtu.dk).

This thesis marks the end of one adventure and the beginning of a new. Enjoy.

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Abstract

An important challenge of Integrated Water Resources Management (IWRM) is to balance water allocation between different users and uses. While economically and/or politically powerful users have relatively well developed methods for quantifying and justifying their water needs, this is not the case for ecosystems – the silent water user. This Ph.D. projects aims at filling the gap by presenting a new environmental flows assessment approach that explicitly links environmental flows, ecosystem services and economic values.

Environmental flows refer to water for ecosystems. Ecosystems, however, provide a wide range of valuable services to people. Therefore, providing for environmental flows is not exclusively a matter of sustaining ecosystems but also a matter of supporting human well being. In the context of IWRM the environmental flows requirement is a negotiated trade-off between water uses. The trade-offs involved are inherently case specific. So are the preferences and policies of decision-makers. In order to facilitate the analysis of trade-offs between various river basin management strategies and water allocation scenarios, environmental flows must be included on equal terms with other water uses. While several holistic and interactive environmental flows assessment methods have been developed, none of them explicitly links environmental flows to ecosystem services. Consequently, such methods cannot readily deliver inputs to economic valuation studies.

This Ph.D. project has developed a simple and transparent decision support tool for assessing various environmental flows scenarios and arriving at a negotiated environmental flows allocation and thereby a negotiated river condition and economic trade-off between water uses. The tool is based on an existing river basin simulation model, MIKE BASIN, and calculation procedures developed in MS Excel. The core of operationalising the tool is the development of the Service Provision Index (SPI). This approach explicitly links environmental flows to (socio)-economic values by deliberately focusing on ecosystem services. As such, it places due emphasis on the 'end product' of ecosystem functions to humans and renders environmental flows somewhat easier to justify and value. Economic valuation of services supported by environmental flows may be done using existing valuation methods. A checklist that links ecosystem services provided by environmental flows to appropriate valuation methods and examples of monetary values is given in this thesis.

While many uncertainties and shortcomings remain, using SPI and economic valuation of environmental flows is a promising way of bringing ecosystems – the silent water user - to the water agenda in IWRM, and it is a novel contribution to the existing field of environmental flows assessment methodologies.

Dansk Resumé

I integreret vandressourceforvaltningen er det en stor udfordring at fordele vandressourcen retfærdigt mellem brugerne. Økonomisk og/eller politisk stærke brugere, såsom landbrug, by-samfund og industri, har udviklet gode metoder til at kvantificere og retfærdiggøre deres behov for vand. Dette er ikke tilfældet for økosystemerne, der risikerer at blive overset, fordi de er ”tavse brugere af vand". Formålet med denne Ph.D. afhandling er at udvikle en metode, hvormed økosystemernes behov for vand kan få en stemme i integreret vandressourceforvaltning.

’Environmental Flows’ refererer til økosystemers behov for vand. Økosystemer bidrager med mange værdifulde ydelser til mennesker. Dermed er ’Environmental Flows’ ikke kun vigtig for opretholdelsen af økosystemerne selv, men også vigtig for menneskers levevilkår. I integreret vandressourceflorvaltning er ’Environmental Flows’ ofte et resultat af forhandlinger mellem stærke og svage brugere af vand samt afvejninger af modstridende hensyn. Hvilke brugere af vand, der findes, afhænger helt af det enkelte områdes karakteristika. Vægtningen af hensynene afhænger til gengæld af beslutningstagernes præferencer og politik. Beslutningstagerne har brug for at kunne vurdere betydingen af forskellige afvejninger og dermed forskellige vandforvaltnings-strategier. I denne vurdering er det vigtigt at ’Environmental Flows’ indgår på lige fod med de andre brugere af vand. Selvom der er udviklet mange holistiske og interaktive metoder til at bestemme ’Environmental Flows’, kan ingen af dem koble ’Environmental Flows’ til økosystemernes ydelser. Derfor kan metoderne ikke bruges til at anslå værdien af ’Environmental Flows’, som netop findes i kraft af disse ydelser.

Denne Ph.D. afhandling har udviklet en simpel metode, der sammenkobler økosystemernes vandbehov, ydelser og værdier ved brug af et såkaldt ’Service Provision Index’ (SPI). Med denne tilgang sættes der fokus på de ydelser, økosystemerne bidrager med til mennesker. Det øger sansynligheden for, at økosystemernes vandbehov kan blive tilgodeset i integreret vandressourceforvaltning. Der findes en lang række økonomiske værdisætningsmetoder, som kan anslå værdien af disse ydelser. I afhandlingen er der udarbejdet en checkliste, der kombinerer økosystem-ydelser med relevante værdisætningsmetoder og eksempelvise økonomiske værdiansættelser. Via MS Excel kan SPI-metoden inkorporeres i en allerede existerende vandressurce-simulationsmodel, MIKE BASIN, hvorved forskellige vandallokerings-scenarier kan blive evalueret med hensyn til både de økologiske og de økonomiske konsekvenser.

Selvom der fortsat eksisterer usikkerheder og ufuldkommenheder, så er SPI-metoden og økonomisk værdisætning af ’Environmental Flows’ en lovende ny fremgangsmåde til at give de ”tavse brugere af vand", økosystemerne, en stemme i integreret vandressourceforvaltning.

Contents

List of abbreviations

AC = Avoided Cost method
BBM = Building Block Method
BT = Benefit Transfer
CBA = Cost-Benefit Analysis
DR = Dose-Response method
CV = Contingent Valuation
DRIFT = Downstream Response to Imposed Flow Transformation
EFA = Environmental Flows Assessment
ES = Ecosystem Services
IFIM = Instream Flow Incremental Methodology
IWMI = International Water Management Institute
IWRM = Integrated Water Resources Management
MCA = Multi-criteria Analysis
ME = Mitigative Expenditure method
MP = Market Price method
PES = Payments for Ecosystem Services
PHABSIM = Physical Habitat Simulation model
RC = Replacement Cost method
RVA = Range of Variability Approach
SPI = Service Provision Index
SS = Service Suitability
TEV = Total Economic Value
WTA = Willingness to Accept
WTP = Willingness to Pay

1 Introduction

1.1 Background

The flows of the world's rivers are increasingly being modified through impoundments such as dams and weirs, abstractions for agriculture and urban water supply, drainage return flows, maintenance of flows for navigation, and structures for flood control (Dyson et al., 2003; Postel & Richter, 2003). These interventions have caused significant alteration of flow regimes mainly by reducing the total flow and affecting the variability and seasonality of flows. It is estimated that more than 60 % of the world's rivers are fragmented by hydrological alterations (Ravenga et al., 2000). This has led to widespread degradation of aquatic ecosystems (Millennium Ecosystem Assessment, 2005).

Globally, there is a growing acceptance of the need to safeguarding ecosystems when managing waters to meet human demands (Instream Flow Council, 2002; Dyson et al., 2003; Postel & Richter, 2003). A goal of Integrated Water Resources Management (IWRM) is to ensure that the efficient use of water and related resources does not compromise the sustainability of vital ecosystems (GWP, 2000; GWP, 2003). This entails finding the balance between the short-term needs of social and economic development and the protection of the natural resource base for the longer term. An important challenge of IWRM is, therefore, to balance water allocation between different users and uses (GWP, 2000). While economically and/or politically powerful users have relatively well developed methods for quantifying and justifying their water needs, this is not the case for ecosystems – the silent water user. Therefore, ecosystems are frequently omitted from water allocation decision-making. Ecosystems, however, provide a wide range of valuable services to people (GWP, 2003; Millennium Ecosystem Assessment, 2005). In developing countries, the livelihood of rural people to a large extent depends directly on the provision of ecosystem services.

The marginalization of ecosystems in water resources management and the associated degradation or loss of ecosystem services, have resulted in economic costs, in terms of declining profits, remedial measures, damage repairs and lost opportunities. The highest costs, however, are typically borne by people depending directly on ecosystem services. These people are generally among the poorest. (Emerton & Bos, 2005; Millennium Ecosystem Assessment, 2005; Pearce et al., 2006).

In several cases, maintaining ecosystems has proven to be a more cost-effective way of providing services than employing artificial technologies (Emerton & Bos, 2005). Thus, recognizing the full value of ecosystem services, and investing in them accordingly, can safeguard livelihoods and profits in the future, save considerable costs and help achieve sustainable development goals. Failing to do so may seriously jeopardize any such efforts (Russell et al., 2001; Costanza, 2003; Dyson et al., 2003; Emerton & Bos, 2005; Millennium Ecosystem Assessment, 2005; Pearce et al., 2006).

Many factors, such as water quality, sediments, food-supply and biotic interactions, are important determinants of riverine ecosystems. However, an overarching master variable is the river's flow regime (Poff et al., 1997, Bunn & Arthington, 2002). The Natural Flow Paradigm (Poff et al., 1997), where the natural flow regime of a river is recognised as vital to sustaining ecosystems, has now been widely accepted (Poff et al., 2003; Postel & Richter, 2003; Tharme 2003). This recognition of flow as a key driver of riverine ecosystems has let to the development of the environmental flows concept (Dyson et al., 2003).

In IWRM, environmental flows serve to represent water allocation for ecosystems. As ecosystems, in turn, provide services to people, providing for environmental flows is not exclusively a matter of sustaining ecosystems but also a matter of supporting humankind/livelihoods, in particular in developing countries. One of the most promising ways of placing ecosystems on the water agenda is by economic valuation of such services. (Millennium Ecosystem Assessment, 2005). In this way ecosystems can be compared to other water using sectors and internalized in decision-making processes.

There is, however, a lack of operational methods to demonstrate the inherently multi-disciplinary link between environmental flows, ecosystem services and economic value. The present Ph.D. project aims at filling this knowledge gap.

1.2 Aim and objectives

The aim of the Ph.D. project is to develop an operational tool for quantifying environmental flows in the context of Integrated Water Resources Management (IWRM).

The objectives are:

1. To review existing methods for quantification of environmental flows and evaluate their applicability in an IWRM context.

2. To compile a checklist of ecosystem services sustained by environmental flows.

3. To review existing economic valuation methods and evaluate their applicability for valuating ecosystem services sustained by environmental flows.

4. Based on MIKE Basin and MS Excel, to develop and apply a simple and transparent decision support system for assessing various environmental flows scenarios and arriving at a negotiated environmental flows allocation.

1.3 Definitions and approaches

The following definition of Integrated Water Resources Management (IWRM) provided by Global Water Partnership (GWP) is adopted in this report:

'IWRM is a process, which promotes the co-ordinated development and management of water, land and related resources, in order to maximize the resultant economic and social welfare in an equitable manner without compromising the sustainability of vital ecosystems' (GWP, 2000).

The definition of Environmental Flows adopted in this report is adopted from Dyson et al. (2003):

'An Environmental Flow is the water regime provided within a river, wetland or coastal zone to maintain ecosystems and their benefits' (adopted from Dyson et al., 2003).

Other definitions and terms regarding environmental flows do exist in the literature. These includes minimum-, in stream- and ecological flow. However, the above definition and the term 'environmental flow' are the only ones truly encompassing the holistic nature of the concept. They are, therefore, adopted in this study.

The *condition,* in which riverine ecosystems and their services are *maintained*, is essentially a socio-political decision. The desired ecosystem condition may be set (e.g. by legislation or international conventions), and the environmental flow requirement is the water regime needed to maintain the ecosystems in that desired condition. Alternatively, the environmental flow allocated to a river system may be a negotiated trade-off between water users. In this case, the resulting ecosystem condition is determined by that negotiated and 'desired' environmental flow.

Setting environmental flows requirements thus may take two fundamentally different approaches depending on the objective in question:

- How much water/flow does a given ecosystem condition need?
- How much water/flow does society allocate ecosystems - and what is the resulting ecosystem condition maintained by this given water/flow allocation? – and is this condition desirable and sufficient?

In the context of IWRM and this Ph.D. project, the latter approach is the most relevant, since it enables (at least in theory) an optimal allocation of the entire water resource among all uses (and allows for adaptive management). The former approach is more rigid, and in this case societal optimisation of water allocation does not include environmental flows, which is fixed (albeit with great uncertainty).

Focus in this study is on end results of ecosystem functioning to humans, in other words, focus is on ecosystem services. As such, it is founded on an anthropocentric ideology.

1.4 Limitations

Environmental flows include water quality as well as water quantity. This report will not address the water quality issue. If there is no water, talking about water quality becomes redundant. Also, while groundwater is an integral part of IWRM and may be important for ecosystems, groundwater is not explicitly included in the present analysis. Neither are factors that may influence service provision in addition to flow. These limitations are necessary in order to stay within the scope and focus of the thesis.

1.5 Outline of thesis

Following this introductory Chapter 1, Chapter 2 presents the history and state of the art for quantifying environmental flows. Existing environmental flows assessment methods are then evaluated with respect to their applicability in an IWRM context. A checklist of ecosystem services sustained by environmental flows is given Chapter 3 in order to provide the missing link between flows and value. Economic valuation is the theme for Chapter 4. The chapter describes the concept of Total Economic Value, reviews existing economic valuation methods and evaluates their applicability in the context of environmental flows assessment. Chapter 5 elaborates on the findings of Chapter 2, 3 and 4 and develops an approach to linking flows, services and values and thereby providing the decision space for quantification of environmental flows in IWRM. The developed approach is applied in East Rapti River Basin, Nepal, and results from this case study are discussed in Chapter 6. Chapter 7 briefly summarizes the papers prepared and based on the preceding chapters, while Chapter 8 provides a summary of the thesis and draws the conclusions.

2 Environmental Flows

2.1 Introduction

The science of environmental flows is relatively new. The development of environmental flows assessment (EFA) methodologies began in USA in the late 1940s and picked up during the 1970s, mainly as a result of new environmental and freshwater legislation accompanying the peak of the dam-building era in USA. Outside the USA, the development of EFA methodologies only gained significant ground in the 1980s or later. Australia and South Africa are among the most advanced countries with respect to development and application of EFAs (Tharme, 2003).

Many early applications of environmental flows were focused on single species or single issues. Much of the demand for environmental flows in North America was from recreational fishermen concerned about the decline in trout and salmon numbers. As a result, environmental flows were set to maintain critical levels of habitat for these species. However, managing flows without consideration for other ecosystem components may fail to capture system processes and biological community interactions that are essential for creating and sustaining the habitat and well-being of that target species.

Since these fish species are very sensitive to flow, it has been argued that if the flow is appropriate for them, it will probably serve most other ecosystem needs. However, a vast body of scientific literature reveals that this may not necessarily be so, and flow management is best addressed for the entire ecosystem. Recent advances in EFAs reflect this knowledge and EFA methodologies increasingly take a holistic approach (Brown & King, 2003, Instream Flow Council, 2002). Also, The Natural Flow Paradigm, where the natural flow regime of a river (comprising the five main components of variability, magnitude, frequency, duration, timing and rate of change) is recognised as vital to sustaining ecosystems, has now been widely accepted (Poff et al., 1997; Postel & Richter, 2003; Tharme, 2003). However, simply mimicking the shape of the natural hydrograph, but at a much lower level, may be none or counter productive (Instream Flow Council, 2002).

A further trend in EFAs is a shift from prescriptive to interactive approaches (Tharme, 2003). The type of approach is closely linked to the objective of the EFA (see 1.3). When clear objectives are defined (e.g. protection of certain species, flooding of specific areas, achievement or maintenance of certain river conditions), a prescriptive EFA recommends a single environmental flow. By using this prescriptive approach, however, insufficient information is supplied on the implications of not providing the recommended flow. Interactive EFAs focus on establishing the relationship between river flow and one or more attributes of the river-system. This relationship may then be used to describe environmental/ecosystem implications (and resulting social/economic implication) of various flow scenarios. Interactive methodologies thus facilitate the exploration of trade-offs of several water allocation options. Interactive approaches may, of course, be used prescriptively.

The basis of most EFAs is a bottom-up approach, which is the systematic construction of a modified flow regime from scratch on a month-by-month (or more frequent) and element-by-element basis, where each element represents a well defined feature of the flow regime intended to achieve particular objectives. In contrast, top-down approaches define the environmental flows requirement in terms of accepted departures from the natural (or other reference) flow regime. Thus, top-down approaches are less susceptible to omission of critical flow features than bottom-up approaches.

In the following, the various assessment methods for environmental flows will be presented and evaluated with respect to their applicability in an IWRM context.

2.2 Environmental flows assessment methods

In the most recent review of international environmental flows assessments, Tharme (2003) recorded 207 different EFA methodologies within 44 countries. Several different categorizations of these methodologies exist, three of which are shown in Table 1.

Table 1 Three different categorizations of EFA methodologies.

Organisation	Categorization of EFA	Sub-category	Example
IUCN (Dyson et al. 2003)	Methods	Look-up tables	Hydrological (e.g. Q95 Index) Ecological (e.g. Tennant Method)
		Desk-top analyses	Hydrological (e.g. Richter Method) Hydraulic (e.g. Wetted Perimeter Method) Ecological
		Functional analyses	BBM, Expert Panel Assessment Method, Benchmarking Methodology
		Habitat modeling	PHABSIM
	Approaches		Expert Team Approach, Stakeholder Approach (expert and non-expert)
	Frameworks		IFIM, DRIFT
World Bank (Brown & King, 2003)	Prescriptive approaches	Hydrological Index Methods	Tennant Method
		Hydraulic Rating Methods	Wetted Perimeter Method
		Expert Panels	
		Holistic Approaches	BBM
	Interactive approaches		IFIM DRIFT
IWMI (Tarme, 2003)	Hydrological index methods		Tennant Method
	Hydraulic rating methods		Wetted Perimeter Method
	Habitat simulation methodologies		IFIM
	Holistic methodologies		BBM DRIFT Expert Panel Benchmarking Methodology

The categorization by IWMI (Tharme, 2003) is the most logical, since it is based on the required biophysical input data and not on the methodological characteristics, which may change over time and be overlapping. This categorization will, therefore, be used in the following brief review of methodologies. The review is based on Tarme (2003), Dyson et al. (2003), Brown & King (2003) and Acreman & Dunbar (2004).

2.2.1 Hydrological Index Methods

These are the simplest and most widespread EFA methods. They are often referred to as desk-top or look-up table methods (see Table 1) and they rely primarily on historical flow records. Environmental flow is usually given as a percentage of average annual flow or as a percentile from the flow duration curve, on an annual, seasonal or monthly basis. Most methods simply define the minimum flow requirement, however, in recognition of the 'Natural Flow Paradigm' more sophisticated methods have been developed that take several (up to 32) flow characteristics into account (such as low-flow durations, rate of flood rise/fall etc).

The most frequently used methods include the Tennant Method (Tennant, 1976) and RVA (Range of Variability Approach) (Richter et al., 1997) both developed in the USA.

Hydrological Index Methods provide a relatively rapid, non-resource intensive, but low resolution estimate of environmental flows. The methods are most appropriate at the planning level of water resources development, or in low controversy situations where they may be used as preliminary estimates.

2.2.2 Hydraulic Rating Methods

These methods were mainly developed and used to recommend in-stream flow requirements of fish in the USA. In recent years, however, they have been superseded by Habitat Simulation Methodologies or absorbed within Holistic Methodologies.

Hydraulic Rating Methods are based on historical flow records and cross-section data in critically limiting biotopes e.g. riffles. They model hydraulics as function of flow and assume links between hydraulics (wetted perimeter, depth, velocity) and habitat availability of target biota. In other words they use hydraulics as a surrogate for the biota. Environmental flow is given either as a discharge that represents optimal minimum flow, below which habitat is rapidly lost, or as the flow producing a fixed percentage reduction in habitat availability.

The Wetted Perimeter Method (Reiser et al., 1989) is the most commonly applied hydraulic rating method.

2.2.3 Habitat Simulation Methodologies

Habitat simulation methodologies are widely used and based on hydrological, hydraulic and biological response data. They model links between discharge, available habitat conditions (incl. hydraulics) and their suitability to target biota. Thus, habitat conditions are directly related to the (predicted) requirements of target species. Environmental flow is predicted from habitat-discharge curves or habitat time and exceedence series.

PHABSIM (Physical HABitat SIMulation model) (Bovee, 1986) is the most commonly applied habitat simulation methodology.

2.2.4 Holistic Methodologies

Holistic methodologies are actually frameworks that incorporate hydrological, hydraulic and habitat simulation models. They are the only EFA methodologies that explicitly adopt a holistic, ecosystem-based approach to environmental flow determinations.

The Instream Flow Incremental Methodology (IFIM) (Bovee, 1986; Bovee et al., 1998), developed in the USA, is the most commonly used and best documented holistic methodology, while the Downstream Response to Imposed Flow Transformation (DRIFT) (King et al., 2003) developed in South Africa, is one of the newest, offering promising and innovative advances to interactive, top-down EFAs. DRIFT has emerged from the foundations of the widely used prescriptive, bottom-up holistic method, the Building Block Method (BBM) (Tharme & King, 1998; King et al., 2000), also developed in South Africa. In Australia, The Holistic Method and the Benchmarking Method (Arthington, 1998), are the most used holistic methodologies, with the latter being the only EFA specifically designed to assess the risk of environmental impacts due to river regulation at basin scale.

BBM and DRIFT are the only two EFA methodologies that consider socio-economic aspects of environmental flows.

Holistic methodologies are believed to be the way forward, and DRIFT is seen as one of the frontrunners of such scenario-based EFA methodologies. DRIFT has great potential for being further operationalised and developed into an IWRM tool.

2.3 Evaluation of existing methodologies

Table 2 summarises the major advantages and disadvantages of using the different methodologies.

Table 2 Major advantages and disadvantages of environmental flow assessment methodologies.

	Duration of assessment (months)	Major advantages	Major disadvantages
Hydrological Index	½	Low cost, rapid to use	Not site-specific, ecological links assumed
Hydraulic rating	2-4	Low cost, site specific	Ecological links assumed
Habitat simulation	6-18	Ecological links included	Extensive data collection and use of experts, high cost
Holistic	12-36	Covers most aspects	Requires very large scientific expertise, very high cost, not operational

Based on various literature reviews (Instream Flow Council, 2002; Postel & Richer, 2003; Tharme, 2003; Dyson et al., 2003; Brown & King, 2003), the following major shortcomings/drawbacks of present EFA methodologies have been extracted:

a. Links between flow and ecosystem functions/components are often assumed and not well documented. This uncertainty is frequently used to argue against meeting recommended environmental flows.
b. Focus is on minimum flow, although safeguarding of variability is equally important (the Natural Flow Paradigm).
c. Focus is on instream/fluvial requirements of riverine systems, while lotic, riparian, floodplain (terrestic), estuarine, and deltaic requirements are often neglected.
d. Relatively little attention is given to the requirements of maintaining morphological processes.
e. Socio-economic aspects are mostly ignored.
f. Validation is difficult, requires long-term monitoring using objectively verifiable indicators.
g. None of the methods have been rigorously tested - there is a need for large-scale experiments.
h. Habitat simulation and holistic methodologies rely heavily on expert judgements.

Bearing these shortcomings in mind, there is obviously a need for improving existing environmental flow methodologies. Although the most urgent and crucial research gap is that of understanding the links between flow and ecosystem functions, the intention of the current research project is not to bridge this gap. Rather, it will build on existing knowledge (a tiny suspension bridge across the gap) and address another hampering shortcoming: the lack of incorporation of socio-economic aspects. Results from existing methods cannot readily deliver inputs to economic valuation studies. Consequently, there is a communication gap between bio-physical disciplines (e.g. ecology, hydrology) and socio-economic disciplines. The key to bridging this gap is to focus explicitly on ecosystem services provided by environmental flows.

While several holistic and interactive environmental flows assessment methods have been developed (Tharme, 2003; Dyson et al. 2003; Brown & King, 2003; Acreman & Dunbar, 2004), none of them explicitly links environmental flows to ecosystem services. Furthermore, existing holistic environmental flows assessment methods are very resource (time, money, data) demanding (ibid.). This is a major constraint for undertaking environmental flows assessments - in particularly in developing countries. Thus, there is a need for developing a holistic 'desktop' environmental flows assessment method that pays due attention to the ecosystem services provided to people (the socio-economic aspect). Ecosystem services are the focus of Chapter 3.

3 Ecosystem Services

3.1 Introduction

In this chapter, focus is on services provided by environmental flows to the benefit of people. This approach renders environmental flows somewhat easier to justify and value, thereby increasing the likelihood of having environmental flows incorporated into decision-making in Integrated Water Resource Management (IWRM).

Ecosystems provide a wide range of services to people (Costanza, 2003; Emerton & Bos, 2005; Millennium Ecosystem Assessment, 2005; Pearce et al., 2006). The services provided by environmental flows, may either be provided directly by flow (e.g. flushing of sediments, salinity control) or indirectly via ecosystem functions (see Figure 1). The extent to which ecosystem functions create ecosystem services depends on the cultural, socio-economic and technical setting. Thus, the list of services given in Table 3 is not entirely determined by the suite of ecosystem functions, but also by human ingenuity in deriving benefits.

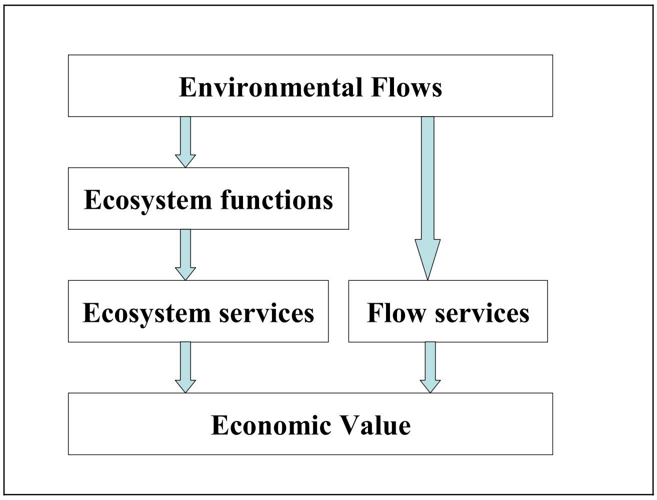

Figure 1 Links between flows, functions, services and value

Bunn and Arthington (2002) have proposed four flow-related key principles that define the influence of flow on aquatic ecosystem functions. *Principle 1*: Flow is a major determinant of physical habitat availability, which in turn is a major determinant of biotic composition. *Principle 2*: Aquatic species have evolved life history strategies primarily in direct response to the natural flow regimes. *Principle 3*: Maintenance of natural patterns of longitudinal and lateral connectivity is essential to the viability of populations of many riverine species. *Principle 4*: The invasion and success of exotic and introduced species in rivers is facilitated by the alterations of flow regimes.

For the purpose of this study it is not necessary to distinguish between direct and indirect services. Therefore, in the following, all services provided by environmental flows will be referred to as ecosystem services.

3.2 Ecosystem services supported by environmental flows

Table 3 is a comprehensive checklist of ecosystem services supported by environmental flows.

Table 3 A complete checklist of all the known possible services provided by natural flow regimes, the flow related functions underlying such provisions and the key components of an environmental flow regime supporting such provisions. The list elaborates on work done by De Groot (1992).

Service category	Service provided	Key flow related function	Key Environmental Flow component or indicator
Production	Water for people - subsistence/rural and piped/urban	Water supply	Floodplain inundataion
	Fish/shrimp/crabs (non-recreational)	Habitat availability and connectivity, food supply	Instream flow regime, floodplain inundation, flows sustaining riparian vegetation
	Fertile land for flood-recession agriculture and grazing	Supply of nutrients and organic matter, moisture conditions in soils	Floodplain inundation
	Wildlife for hunting (non-recreational)	Habitat availability and connectivity, food supply	Floodplain inundation, flows sustaining riparian vegetation
	Vegetables and fruits	Supply of nutrients and organic matter, seasonality of moisture conditions in soils	Floodplain inundation, flows sustaining riparian vegetation
	Fibre/organic raw material for building/firewood/handicraft	Supply of nutrients and organic matter, seasonality of moisture conditions in soils	Floodplain inundation, flows sustaining riparian vegetation
	Medicine plants	Supply of nutrients and organic matter, seasonality of moisture conditions in soils	Floodplain inundation, flows sustaining riparian vegetation
	Inorganic raw material for construction and industry (gravel, sand, clay)	Sediment supply, transportation and deposition (fluvial geomorphology)	Instream flow magnitude and variability
Regulation	Chemical water quality control (purification capacity)	Denitrification, immobilization, dilution, flushing,	Floodplain inundation, instream flow regime,
	Physical water quality control	Flushing of solid waste, flushing/retention of sediment, shading	Floodplain inundation, instream flow regime, flows sustaining riparian vegetation
	Flood mitigation	Water retention capacity	Floodplain inundation, flows sustaining riparian vegetation
	Groundwater replenishment (low flow maintenance)	Groundwater (aquifer) replenishment	Floodplain inundation
	Health control	Flushing of disease vectors	Instream flow regime, water quality
	Pest control	Habitat diversity, disturbance and stress	Instream flow regime
	Erosion control (riverbank/bed and delta dynamics)	Healthy riparian vegetation, erosion, transportation and deposition of sediments	Flows sustaining riparian vegetation
	Prevention of saltwater intrusion (salinity control)	Freshwater flow, groundwater replenishment	Instream flow regime
	Prevention of acid sulphate soils development	Groundwater replenishment	Floodplain inundation
	Carbon "trapping" (sequestration)	Accumulation of organic material in peat soils	Floodplain inundation

	Microclimate stabilization	Healthy ecosystems	Floodplain inundation, flows sustaining riparian vegetation
Information	Recreation and tourism (incl. fishing and hunting)	Presence of wildlife, aesthetic significance, good water quality	Site specific
	Biodiversity conservation	Sustaining ecosystem integrity (habitat diversity and connectivity)	Natural flow regime
	Cultural/religious/historical/symbolic activities	Site specific	Site specific
Life support	The prior existence of healthy ecosystems	All	Natural flow regime

In the first column four different service categories are presented: production, regulation, information and life-support. Production services refer to products provided by ecosystems, regulation services are benefits obtained from the regulation of ecosystem processes, information (or cultural) services are the nonmaterial benefits obtained from ecosystems, while life-support services are those that are necessary for the provision of all other services. It is important to note that the service categories should not be considered exclusive or independent. The service categories largely correspond to the value categories presented in Chapter 4.2. This is shown in Paper A and applied in Paper C.

The second column in Table 3 shows the services provided within each service category. Answering the question: 'what's the link with flow?' is indispensable when talking about environmental flows. Therefore, the third column attempts to answer this question by suggesting flow-related key functions supporting the provision of services. It is acknowledged that other functions and conditions influence the provision of services, but addressing them is beyond the scope of this project. Finally, column 4 summarizes the key component or indicator of environmental flows that must be included in an environmental flow assessment, if provision of the related service is to be evaluated and subjected to an economic analysis.

Previous lists have focused on ecosystem functions, e.g. De Groot et al. (2002), but Table 3 focuses on the services. This is to enable the link to economic value. De Groot et al. (2002) include a category called 'habitat functions'. In Table 3 the provision of habitats (incl. breeding area and migration 'rest-place') is included in other services. For example, if the habitats produce fish that are consumed, habitat provision is included in production service. If habitats support other services (e.g. recreation), the value is included in these services. If habitats provide no direct or indirect value, they are included in biodiversity conservation.

3.3 Criteria for selecting ecosystem services

The benefits of ecosystems services can be far removed in time and space from the ecosystem that provides them. Ideally, all ecosystem services supported by environmental flows should be included in the environmental flows assessment. In reality, the resources available for undertaking such an assessment will often be limited and only the most important services can be subjected to further analysis. In such cases, clear selection criteria must be defined. The appropriate criteria for selecting important ecosystem services depend entirely on the objectives of the environmental flows assessment and thus on the political issues addressed and prioritized. This frames/scopes the assessment and defines the spatial, socio-economic and temporal scales.

Spatial scale refers to the geographical extent of the services to be considered. Providing for environmental flows in one river basin may support services further downstream. For transboundary rivers this is a crucial issue that places environmental flows in the centre of the 'payment for ecosystem services (PES)' discourse. Within a given spatial unit, ecosystem services play different roles in people's livelihood strategy. This may be termed the socio-economic scale. Ecosystem services may also produce socio-economic secondary spin-off effects, such as supporting social structures and employment, and preventing pauperisation and conflicts. Such spin-off effects are not included in Table 3 above (it includes only primary services), and the extent to which they are included in the further analysis must be defined. A considerable 'time lag' may elapse before changes in ecosystems manifest themselves. Therefore, not only existing but also potential ecosystem services must be considered.

Once the scale issues have been resolved, identification of important services should be a participatory process that allows all stakeholder/beneficiary groups to be involved. This could be done by showing Table 3 (in a simplified, preferably visualized form) to stakeholders and asking them to select the services they are aware of and find most important. Some of the more intangible (or 'large scale') services, for example carbon sequestration and biodiversity conservation, do not have clearly defined beneficiaries and experts may be needed to identify such services.

The benefits to people of providing for environmental flows, and thus sustaining ecosystem services, are multifaceted. One way of enabling the comparison of benefits and the evaluation of scenarios is by economic valuation of ecosystem services. Such economic valuation is the focus of the following Chapter 4.

4 Economic Values

4.1 Introduction

As shown on Chapter 3, ecosystems provide a wide range of services to people. One of the most promising ways of placing ecosystems on the water agenda is by economic valuation of services sustained by ecosystems. In this way ecosystem services can be compared to those in other sectors and internalized in decision-making processes. Therefore, focus in this Chapter 4 is on the economic valuation of ecosystem services provided by environmental flows.

Economic valuation of ecosystem services is fundamentally rejected by ecocentric environmentalists who argue that humans are not capable of setting a price on ecosystems (Turner et al., 1994; Costanza, 2003). Furthermore, while some argue that existing valuation methods are mature and capable of providing useful information (National Research Council, 2005), others have criticized existing methods claiming they are inadequate and misleading (Merrett, 2005). But as long as we are making choices that affect ecosystems, we are doing valuation of ecosystems, whether acknowledged or not. It is of utmost importance that this valuation is made explicit in order to ensure a comprehensive basis for decision-making including high levels of information and transparency.

Economic valuation aims at quantifying the contribution of resource use (including ecosystem services) to human well-being. This is done by measuring or inferring human preferences. As such, economic valuation of ecosystems is founded on an anthropocentric, utilitarian ideology with the goal of maximizing individual (or societal) utility (Turner et al., 1994; Bockstael et al., 2000; Farber et al., 2002). It has been argued, however, that other goals, for instance that of sustainability or social equity, should be considered (Limburg et al., 2002; Farber et al., 2002; Constanza, 2003; Newcome et al., 2005). For the purpose of informing decision-making in a real world context, economic valuation is the most relevant and well developed concept available (Pearce at al 2006). Also, economic valuation of ecosystems serves several other purposes than attempting to internalize externalities and secure efficient decisions. It places ecosystems services on various practical policy-making agendas: poverty reduction, sustainability, equity etc.

There are two important assumptions underlying economic valuation: marginality and substitutability. The change to be valued must be marginal. If attempting to do economic valuation based on a non-marginal change, the exercise may be meaningless and/or alter the unit being used in valuation (e.g. existing market prices may be affected and cannot be used for valuation). The utilitarian principle of substitutability implies that all values (types of capital) are substitutable or replaceable. This is indicative of the so-called weak sustainability approach. Both assumptions are critical challenges for valuating ecosystem services in developing countries, see Paper C for a thorough discussion.

In the next section, the concept of Total Economic Value will be presented. Then follow a review of existing valuation methods and an evaluation of their applicability in the context of environmental flows assessment. Paper A provides a comprehensive checklist linking ecosystem services to type of value and appropriate valuation method. A review of economic values of ecosystem services is given in Paper C.

4.2 Total Economic Value

Basically, the values associated with ecosystems can be divided into two types: use and non-use (or passive-use) values. Most of these values can, albeit not always easily, be monetized to constitute the total economic value (TEV) of ecosystems. TEV of ecosystems can be divided into five categories (see Figure 2): Direct and indirect use, option, bequest (incl. altruism) and existence (Turner et al., 1994). Direct use values are associated with direct use of ecosystem services, such as fishing, hunting and swimming. Indirect use values refer to services like flood mitigation and carbon sequestration that are not directly consumed, but still creates benefits to the current generation. The value of preserving an ecosystem for potential future use by the current generation is termed option value. Non-use values comprise bequest value and existence value. Bequest value is the value that the current generation places on preserving ecosystems for coming generations. The current generation may appreciate the very existence of certain ecosystem assets, such as the blue whale, without any intentions of ever using it (e.g. for recreation). This non-use value is captured by existence value. For a list of ecosystem services contributing to the various types of values, see Paper C.

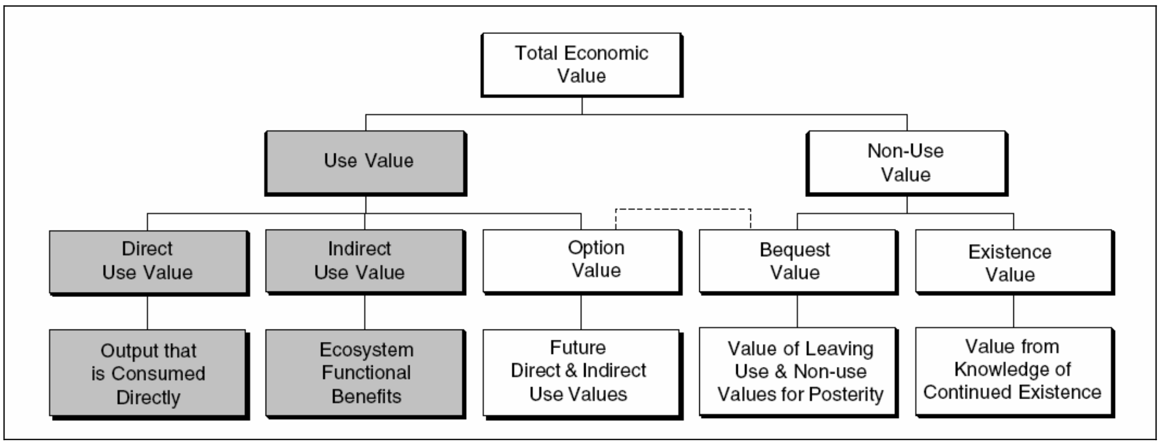

Figure 2 Total Economic Value comprising several use and non-use values. Source: Turner et al 1994.

By definition, TEV is anthropocentric and reflects the preferences (individual or societal) of human beings. It is, therefore, argued that TEV ignores an intrinsic value residing in ecosystems, independently of human preferences. Whether or not an intrinsic value exists is a matter of belief. If it exists it cannot be empirically quantified by humans and has no operational value. The existence value of TEV may capture parts of the intrinsic value and is sometimes termed the anthropocentric intrinsic value (Turner et al., 2003). Often, however, existence value and intrinsic value are incorrectly used as synonyms (Emerton & Bos, 2005; National Research Council, 2005).

TEV is normally calculated as the sum of ‘all’ individual ecosystem services. But ecosystem services may be non-additive, and simply adding their values may underestimate the ‘true’ value of ecosystems (Bockstael et al., 2000). In other words, healthy ecosystems are a prerequisite for the provision of all other services, and thus can be said to possess a monetary value. TEV may fail to fully encompass this overarching life-support service of ecosystems. On the other hand, there is a risk of double-counting and thus overestimating TEV, if individual services overlap (De Groot et al., 2002).

Despite the above mentioned shortcomings, TEV provides a logical and structured approach to valuing ecosystems, and it is considered the most useful approach currently available for undertaking valuation studies. (National Research Council, 2005; Newcome et al., 2005; Pearce et al., 2006).

In a theoretical context, the terms Use and Non-Use values, as introduced above, are intuitively the most appropriate. When it comes to practical application and actual valuation, it is more useful to distinguish between Marketed and Non-Marketed Values. Note that these terms are not synonymous, as both Use and Non-Use values each may include Marketed and Non-Marketed values.

4.3 Economic valuation methods

Several methods have been developed to quantify the various components of the total economic value of ecosystem goods and services. Two basic approaches can be distinguished (Turner et al., 1994):

- Demand Curve approaches
- Non-Demand Curve approaches (cost-based approaches and others)

Demand-curve approaches can be further classified into stated or revealed preference approaches (ibid). The former is based on behavioural intentions, while the latter is based on actual behaviour. Figure 3 shows a classification of the most commonly used methods.

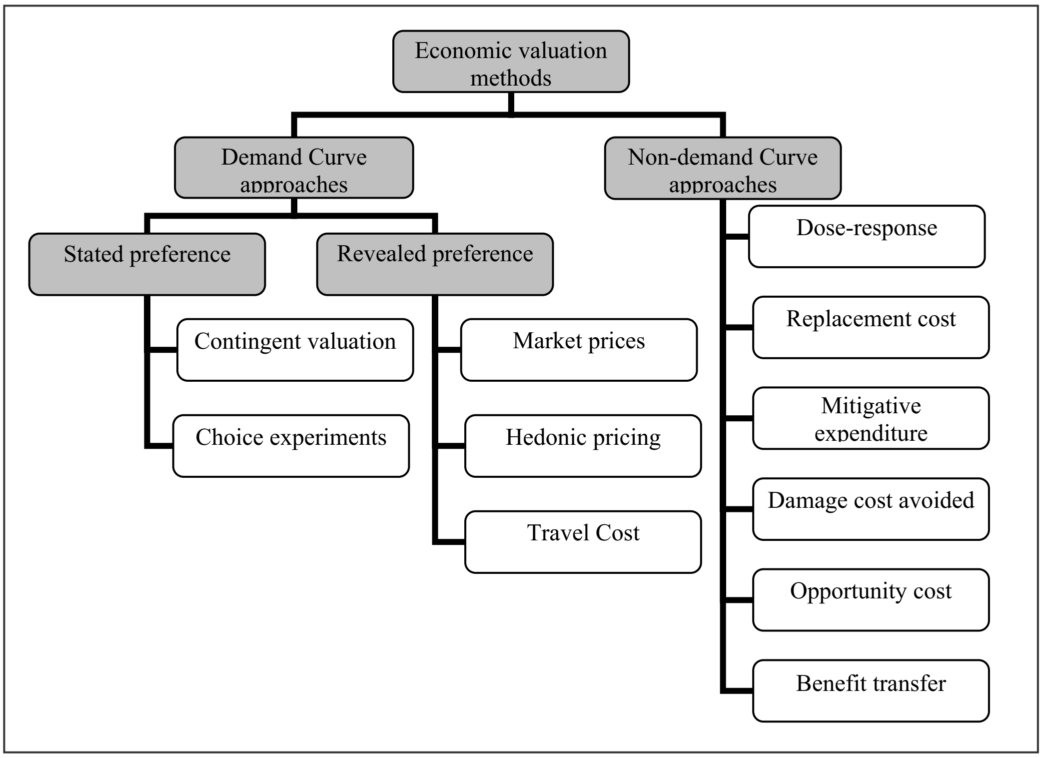

Figure 3 Economic valuation methods.

In the following, each of these methods will be briefly described. The descriptions are mainly based on Turner et al. (1994), Emerton & Bos (2005) and Pearce et al. (2006).

4.4 Revealed preference methods

4.4.1 Market prices

This method uses existing market prices to estimate direct use values of ecosystem services. In theory, this method is applicable to any ecosystem service that produces a product, which can be freely bought or sold. It is easy to use and requires a minimum of data-collection and analysis.

Valuation technique

- Quantify the product
- Find its market price
- Multiply quantity by price

Disadvantages

While the method is relatively easy to use, it has some major disadvantages.

- Market failure: existing markets are distorted and irregular (subsidies, market interventions, non-competitive, imperfect/assymetrical information)
- It is difficult to quantify the product (opportunistic, high levels of substitution and complimentarity)
- Market prices do not necessarily reflect values of services to society, nor the actual willingness to pay (related to market failure)
- Requires access to market

4.4.2 Hedonic pricing

The presence, absence or quality of ecosystem services may influence the market price of other goods and services. Hedonic pricing attempts to value ecosystem services by quantifying this influence. The method has been most commonly applied to the property market, e.g. estimating use values.

Valuation technique

- Determine all the various attributes, including ecosystem goods and services, influencing the market price of a property
- Collect data on property prices in areas with varying quantity and quality of ecosystem services
- Factor out (by statistical analysis) the influence of ecosystem services
- Derive demand curves relating quantity/quality of certain ecosystem services to changes in property prices

Disadvantages

- Requires large and detailed data-sets
- Difficult to isolate specific ecosystem effects from other effects
- Assumes that people have the opportunity to freely select a property within the constraints given by their income (e.g. no limited supply)
- Assumes private ownership

4.4.3 Travel cost

Travel cost methods assume that the incurred cost (including both direct costs and cost of time spend) of visiting a recreational site reflect the minimum recreational value of that site. Travel cost methods are a common way of estimating direct use values (recreational values).

Valuation technique

- Define the total 'catchment area' of a recreational site and divide it into zones of approximately equal travel costs
- Within each zone, sample visitors to collect information about their incurred cost, frequency and motives of the visit, site attributes and socio-economic characteristics
- Obtain the visitation rate
- Estimate travel cost by including both direct costs and time spend on the visit
- Test the relationship between visitation rate and explanatory factors such as travel cost and socio-economic variables
- Construct a demand curve relating visitation rate to travel cost and calculate consumer surplus = value

Disadvantages

- It depends on large and detailed datasets that are expensive to collect (interviews and questionnaires)
- The analytic techniques are relatively complex
- It is very difficult to price the time spent on the visit. Alternative cost methods are often applied. However, to some people the time spent on traveling is not perceived as a cost but a benefit
- Multiple destinations/motives make it hard to separate value of a single site
- People that live close to a site and incur no or very limited travel costs may put a very high value on the site → value to local people may be underestimated

4.5 Stated preference methods

Stated preference methods do not require a market to exist. Nevertheless, it is a prerequisite that respondents are familiar with the concepts of a market and are capable of dealing with trade-offs within their budget constraint.

4.5.1 Contingent valuation

In contingent valuation, a hypothetical market is created. Individuals are asked explicitly to state their willingness to pay (WTP) for ecosystem services or their willingness to accept compensation (WTA) for the loss of ecosystem services. This information, the stated preferences, is used to establish a demand curve or a point on a demand curve. Contingent valuation is widely used to determine both use and non-use values of ecosystems and is one of the only methods available for estimating option-, bequest-, and existence values.

Valuation technique

- Conduct interviews or postal surveys (dichotomous choice or open-ended) to find WTP/WTA for a particular ecosystem service
- Find relationship between WTP/WTA and respondents socio-economic characteristics
- Estimate value placed on the specified ecosystem service by respondents

Disadvantages

- Results may be very biased due to several complications: Part-whole bias, strategic bias (speculative respondents - free ride or warm glow), information bias, payment vehicle bias, starting point bias
- WTP<WTA
- Data collection and analysis are complex

Some ecosystems in developing countries attract significant funding from bilateral and multilateral donors. This could be taken as a WTP by the international community (National Research Council, 2005; Pagiola et al., 2004).

4.5.2 Choice experiments

As in contingent valuations, choice experiment (or conjoint analysis) methods create a hypothetical market. Choice experiments involve presenting several ecosystem scenarios each described by a set of attributes including a price/cost. Respondents are then asked to choose (or rank/rate) their preferred option and thus indicate the trade-offs they are willing to make. Based on this information on stated preferences, a demand curve is established. Choice experiments can thus be used to determine both use and non-use values.

Valuation technique

- Conduct interviews or postal surveys to find preferred trade-off
- Find relationship between trade-off choices and respondents socio-economic characteristics
- Estimate value placed on the ecosystem service by the group of people in question

Disadvantages

- Difficult to design surveys correctly (quantification of scenario attributes, level of information etc.)
- Data analysis is very complex

4.6 Non demand curve approaches

4.6.1 Dose-response

A wide range of ecosystem services are used as basic inputs to or prerequisites for the provision/production of other goods and services. The 'dose-response' method assesses the effect of changes in quality/quantity of ecosystem services on the profitability/size of related productions/outputs. When only addressing the impact on marketed (commercial) production, this method is sometimes referred to as 'effect on production' as both production cost and output quantity/quality may be affected. Similarly, when addressing effects on income the method is referred to as 'factor income'. These relatively simple methods are commonly used and have applicability to a wide range of ecosystem services.

Valuation technique

- Determine and quantify links between ecosystem services and the related output (dose-response relationship)
- Relate a specific change in ecosystem services to a change in output
- Estimate the value of the resulting change in output (by using any valuation method, e.g. market price)

Disadvantages

- It can be difficult to establish a correct dose-response relationship
- Influences of general trends and exogenous factors must be isolated and eliminated or assumed unattached

4.6.2 Shadow price approaches

There are four related, but distinct, methods using a shadow price (or cost-based) approach: replacement cost, mitigative expenditure, damage cost avoided and opportunity cost. In order to demonstrate the differences between the methods, an example is given. Consider the wetland service of flood attenuation capacity. *Replacement cost* involves estimating the cost of creating a storage capacity similar to that of the wetlands. *Mitigative expenditure* estimates the cost of building dykes, widening and deepening channel, pumping water etc. *Damage cost avoided* estimates the cost of lost/damaged agricultural production, infrastructure, settlements etc. due to increased flooding. *Opportunity cost* sets a benchmark value of the lost wetland equal to the value of the development that replaces the wetland. The methods are further described below.

4.6.2.1 Replacement costs

It is sometimes possible to replace or restore ecosystem services with artificial or man-made products, infrastructure or technology. The cost of doing this is an indicator of the value of the ecosystem service, as the expenditure can be seen as an estimate of the WTP for maintaining that ecosystem service. It is a relatively simple and straightforward method that is particularly useful for estimating indirect use values of ecosystem.

Valuation technique

- Identify a possible alternative or substitute for an ecosystem service that gives an equivalent level of benefits to the same population (= a shadow project)
- Calculate the cost of establishing and maintaining this alternative or substitute

Disadvantages

- Difficult to find perfect replacement → undervaluation
- The reality: would such replacement costs/expenditures be considered worthwhile? → overvaluation

4.6.2.2 Mitigative expenditures

When an ecosystem service is lost, it often has negative effects on other economic or 'subsistence' sectors. Such negative effects can in some cases be mitigated by investing in mitigative projects. These mitigative expenditures can be seen as indicators of the value of maintaining the ecosystem service in terms of costs avoided. When the mitigative expenditure is based on demand curves (individual preferences) this method is referred to as averting behaviour.

The method is complimentary to the replacement cost method. While replacement cost is an estimate of the cost of providing similar services as those previously provided by ecosystems, mitigative expenditures are an estimate of the costs of mitigating the loss of an ecosystem service.

As is the case with replacement cost methods, the mitigative expenditure method is relatively simple and straightforward and is particularly useful for estimating indirect use values.

Valuation technique

- Identify the negative effects (type, spatial and temporal distribution) caused by the loss of a particular ecosystem service
- Obtain information on mitigative behaviour and projects
- Cost all the mitigative expenditure

Disadvantages

- Difficult to mitigate perfectly → undervaluation
- The reality: would such mitigative behaviour take place - or would people accept a decrease in benefit/utility/profit? → overvaluation

4.6.2.3 Damage cost avoided

This method attempts to estimate the negative economic impact of loosing an ecosystem service. The value of the service is then assumed equal to the damage cost avoided by maintaining the service.

Valuation technique

- Identify the negative effects (type, spatial and temporal distribution) caused by the loss of a particular ecosystem service
- Obtain information on frequency of damaging events occurring under different scenarios of ecosystem service loss, spread of impacts and magnitude of damage caused
- Estimate the cost of these damages and ascribe the contribution of ecosystem services towards minimizing or avoiding them

Disadvantages

- Difficult to create links between ecosystem services and damage avoided (relate damages to changes in service)
- Damages may be far removed in time and space → under-valuation

4.6.2.4 Opportunity cost

If an ecosystem service is replaced by some other benefit, it can be assumed that the decision-maker - acknowledged or not - ascribed a lower value to the ecosystem service than to the new benefit. This implies that if a cost-benefit analysis been undertaken, the opportunity cost (the value of forgone benefits from ecosystem services) would have been lower than the value of new benefits from development. As such, the opportunity cost approach sets a benchmark for which value the ecosystem service must attain in order for the development not to be worthwhile. In other words, it expresses the maximum perceived value of the ecosystem service.

This method can only be applied for *ex post* valuation. In the case of *ex ante* valuation, results of opportunity cost methods can be used in benefit transfers (see below).

Disadvantages

- Decisions are seldom based on economic considerations only
- Information on the economic value of ecosystem services are sparse → decisions are taken on a biased basis
- Non-marketed values are rarely considered
- As development normally replaces a full suit of ecosystem services, it is difficult to value specific services

4.6.3 Benefit transfer and meta-analysis

Benefit transfer is the use of valuation estimates obtained (by any method) in one study to estimate values of ecosystem services in a different study. Due to low cost and time requirements, this method is attractive and has been widely used. Benefit transfers can either be simple value transfers or more complex function transfers (National Research Council, 2005). In the latter case, meta-analysis using multivariate statistical methods can be used to derive such function by linking value to a set of explanatory variables.

Disadvantages

- Unless the transfer is well justified (e.g. the two contexts are comparable) or appropriately adjusted (using context specific data) benefit transfer may produce results that are very poor (have large transfer errors)

4.7 Evaluation and overview of economic valuation methods

Table 4 is a brief summary of the valuation methods described in the previous chapter.

Table 4 Summary of valuation methods. ES = Ecosystem Service.

Method	Approach	Application	Data requirement	Main limitations
Market prices (MP)	Market prices	Marketable products	Low	Imperfect and inaccessible markets
Hedonic pricing (HP)	Effect of ES on price of other goods	Scenic beauty	High	Assumes freedom to select, difficult to isolate effect of ES
Travel cost (TC)	Demand curve based on actual travel cost	Recreation	Medium	Multiple destinations, people in the vicinity may place high value
Contingent Valuation (CV)	WTP/WTA	Any ES	High	Many biases, difficult to use in a subsistence context
Choice experiments	Preferred scenario	Any ES	High	Same as above
Dose response (DR)	Effect of ES on production of other goods and services	Any ES	Medium	Lack of knowledge about relationship between ES and production
Replacement cost (RC)	Cost of replacing lost ES	Any ES	Medium/ Low	Imperfect or unfeasible replacements
Mitigative expenditure (ME)	Cost of mitigating effects of lost ES	Any ES	Medium/ Low	Imperfect or unfeasiblemitigations
Damage cost avoided (DC)	Damage cost avoided by maintaining ES	Any ES	Medium/ Low	Lack of knowledge about links between ES and damage avoided
Opportunity cost (OC)	Value of development that has replaced ES	Any ES	Low	Doesn’t yield the full value, doesn’t allow for different scenarios
Benefit transfer (BF)	Transfers results of existing valuation studies	Any ES	Very low	Gives poor results if contexts differ

‘Market price’ is the most widespread method used for valuating marketed ecosystem services, also in developing countries. Here, market distortions and limited access to markets are major problems when using this method, not to mention the fact that most services are non-marketed. ‘Travel cost’ is often applied to estimate recreational values. The main point of concern, when applying this method in developing countries, is that the value to local people may be underrated.

‘Stated preference’ methods are the preferred methods for valuating non-marketed services. However, such methods require people to be familiar with the concept of money. In relation to *ex ante* valuation, it is difficult for people to value trade-offs they have not personally experienced. Furthermore, the budget constraint (ability to pay) of poor people can be inhibitory to any realistic expression of value. Hence the preferences of wealthy people may get a higher weight than that of poor people (Merrett, 2005; Pearce at al., 2006). Consequently, ‘stated preference’ methods are problematic in the context of developing countries (large socio-economic scale) and subsistence use.

‘Shadow price’ approaches have been heavily criticised, but are widely used. National Research Council (2005) concludes that replacement cost methods are ‘not valid approaches and should not be employed to value aquatic ecosystem services’. Nevertheless, the same authors include mainly cases using replacement cost (7 out of 14) in their review. ‘Benefit transfer’ is the easiest method to use, and this is reflected in its extensive application (Herman et al., 2006).

In summary, economic valuation does not attempt to come up with a definite, universal value of ecosystems, but merely approximates the contribution of ecosystem services to human well-being. In developing countries, many rural people’s livelihoods depend directly on the provision of ecosystem services. Often, these people are poor and they have few alternatives should the ecosystems deteriorate. In such situations, economic valuation of ecosystem services becomes particularly challenging. The selection of which valuation method to use depends on the services to be valued, the data availability and time constraints.

Despite the shortcomings of every economic valuation method they have one significant virtue in common: they hold great potential for raising awareness about the roles and values of ecosystem services for human well-being. In the following Chapter 5, this potential will be put into play in the context of Integrated Water Resources Management (IWRM) and environmental flows assessment. Linking environmental flows, ecosystem services and economic values is the focus of Chapter 5.

5 Linking Flows, Services and Values

5.1 Introduction

In the context of IWRM, the environmental flows requirement is a negotiated trade-off. In order to facilitate the analysis of trade-offs between various river basin management strategies and water allocation scenarios, environmental flows must be included on equal terms with other water uses. As concluded in chapter 2, there is a need for developing a holistic desktop environmental flows assessment (EFA) method that pays due attention to the ecosystem services provided to people (the socio-economic aspect of EFA). Chapter 3 presented a checklist of such ecosystem services related to environmental flows, while Chapter 4 articulated that economic valuation of services supported by environmental flows is a promising way of bringing environmental flows to the decision-making agenda on equal terms with other water uses.

This chapter describes the development of a simple and transparent decision support tool for assessing various environmental flows scenarios and arriving at a negotiated environmental flows requirement/allocation and thereby a negotiated river condition and economic trade-off between water uses. A concept for the decision support tool is outlined in Paper A. In the following, the tool will be described, and important issues, such as decision-making processes, stakeholder involvement and uncertainties will be discussed.

5.2 The concept: Linking MIKE BASIN, SPI and MS Excel

Figure 4 gives a conceptual overview of the tool. MIKE BASIN is an ArcGIS based river basin simulation model and the Service Provision Index (SPI) is an Environmental Flows assessment approach. MS Excel is used to calculate economic values and explore trade-offs. The resulting tool can serve to support decision-making in IWRM.

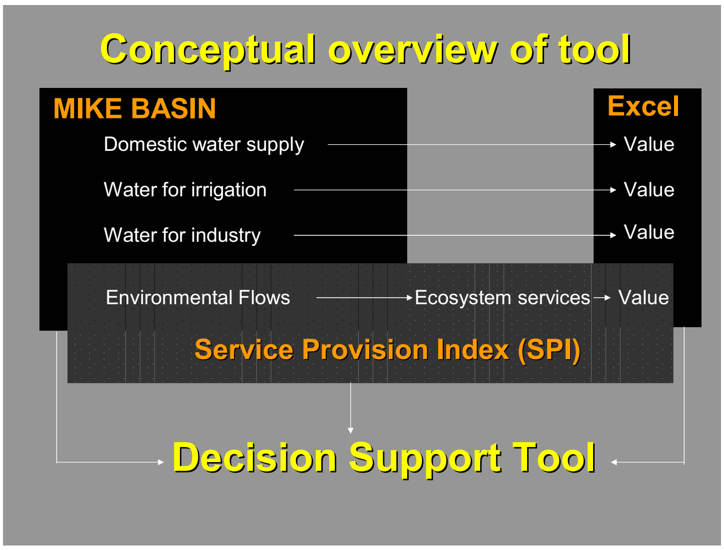

Figure 4 Conceptual overview of how MIKE BASIN (a river basin simulation model), MS Excel and the Service Provision Index (SPI) are linked to provide a decision-support tool for IWRM.

5.2.1 The Service Provision Index (SPI)

The core of operationalising the tool is the development of the Service Provision Index (SPI). This novel approach to assessing environmental flows is described in details in Paper D. In the following, SPI will be briefly presented and main advantages and disadvantages will be discussed.

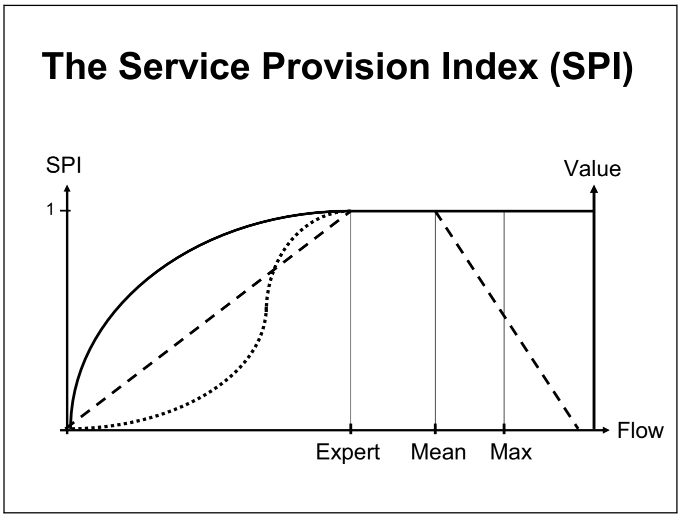

Figure 5 The Service Provision Index (SPI). SPI shows how suitable a given flow scenario is for providing a given service. The SPI can be linked to value. Note that the second y-axis (Value) may or may not be linear. *Expert* refers to a flow lower than mean flow but judged by experts to be sufficient for full service provision (SPI = 1).

For a given flow scenario, the SPI gives a relative estimate of the level of service provision for selected services. Depending on the resources available, the SPI curve may be based on extensive and comprehensive field work (e.g. using the DRIFT framework; King et al., 2003), by using existing species-level information (e.g. PHABSIM; Bovee et al., 1998) or by assuming a certain relationship. In the latter case, the SPI curve may be based on standard linear, logarithmic or logistic relationships (see Figure 5). SPI may decrease if flows are above a certain limit.

In order to establish the SPI curve, appropriate flow classes must be identified for each selected service. An appropriate flow class is any characteristic of the natural flow regime that is considered vital for provision of a particular service. Thus a flow class may be a seasonal mean flow, a particular flood event or minimum flow during a certain period. The number of important flow classes to include in the assessment depends entirely on the service in question. In the case of limited data and knowledge, mean monthly flows can be used by default.

The level of service provision may then be assigned an economic value using one or more existing economic valuation method. (See Chapter 4, Paper A and Paper C). The steps needed in order to use SPI for environmental flows assessment are summarised in Table 5.

Table 5 Overview of steps required to use the Service Provision Index (SPI) method for assessing environmental flows. * These steps may be omitted, if economic valuation of the service provision is not undertaken.

Phase	Step	Comment
Linking flows to services	Identifying all flow related ecosystem services (existing and potential)	Use checklist provided by Paper A and/or framework developed by Meijer (2006)
	Selecting the most important flow related ecosystem services	Should be a stakeholder-oriented and participatory process
	Defining most important flow classes for each service	List of recommended/suggested flow components is a crucial research need
	Quantifying links between flow and each services	Use standard curves or suitability curves based on comprehensive assessments
	Calculating the Service Provision Index (SPI) for each sercice	For a given environmental flows scenario
Linking services to values	Defining the spatial and temporal scale of valuation	Whose benefits should be included?
	*Estimating, for each service, the economic value at a certain SPI	Use existing valuation methods, see Paper C
	*Calculating the economic value of each service	For a given environmental flows scenario
Evaluating environmental flows scenarios	*Calculating total value of each scenario	If economic valuation is undertaken
	Calculating total SPI of each scenario	If economic valuation is not undertaken, total SPI can act as an indicator of the relative value of environmental flows scenarios

The main advantage of the SPI approach is that it explicitly links environmental flows to (socio)-economic values by deliberately focusing on ecosystem services. Furthermore, when establishing the links, a wide variety of information can be used, depending on the resources (time, money, expertise) available for the assessment. This flexible nature is particularly appealing in data-scarce cases and/or in the context of adaptive management.

The SPI approach resembles that of DRIFT (see section 2.2.4) and, in principle, SPI is based on the same structured framework as DRIFT. As such, the SPI approach may be seen as a ‘desktop DRIFT’ with focus on the socio-economic module. This approach differs from existing holistic environmental flows assessment methodologies in several ways. Firstly, while existing methodologies focus on ecosystem components (e.g. fish, invertebrates, plants, water quality, geomorphology), SPI focuses on services – the end product of ecosystem functioning to humans. This is crucial for enabling the subsequent valuation of environmental flows. Secondly, while existing methodologies operate with a fixed number of flow classes (e.g. dry-season low-flows, wet-season low-flows, and eight different flood events), SPI allows a flexible inclusion of the most relevant flow classes. Thirdly, history is not taken into account in existing methodologies. The SPI approach does to some extent allow preceding events to influence the calculation of SPI and corresponding value. Fourthly, existing holistic methodologies are very resource intensive and may take several years with inputs from numerous experts. Depending on the resources available, SPI can be set up from a desk-top study, using standard relationships, or a comprehensive field study. Finally, as SPI is set up in MS Excel it can be easily incorporated into existing river basin simulation models (for example MIKE Basin) and used directly in decision support systems. This possibility of mainstreaming environmental flows into river basin management is a great advantage of the SPI approach.

The main shortcoming of the SPI approach is that in data scarce applications the links between flows and services are assumed. As more and more information becomes available, such links can be refined and documented. Ideally, output from existing holistic environmental flows assessment may directly feed into an SPI for some services (e.g. 'biodiversity conservation'). However, a major challenge remains regarding the establishment of links between flows and services: identifying the extent to which flow is responsible for service provision. For each service, this should be further explored, and empirical guidelines should be developed.

5.2.2 MIKE BASIN

MIKE BASIN is an ArcGIS based river basin simulation model. It is a network model in which branches represent rivers and nodes represent confluences, bifurcations, and locations of water in-takes/out-lets (Figure 6).

MIKE BASIN's computational core can be accessed programmatically, for example by Visual Basic macros from MS Excel. Thus, MIKE BASIN can be run directly from MS Excel, where inputs to and outputs from the model can be processed. This is a powerful characteristic of MIKE BASIN and forms the backbone of the decision support tool developed here.

Depending on the available data, MIKE BASIN is set up using either existing flow data or rainfall-runoff modelling (DHI, 2005). Each 'conventional' water user is primarily defined by a location and a water demand time series. Wetlands may be represented by shallow reservoirs and in this case some state variables must be given.

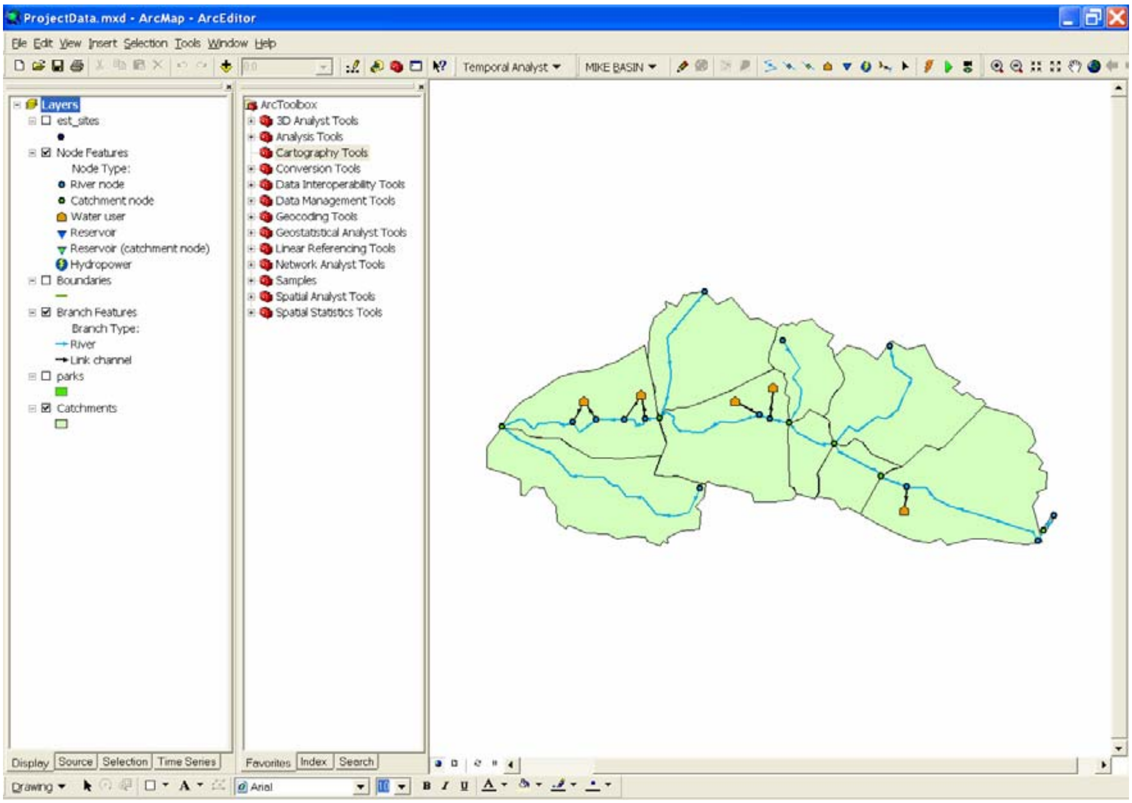

Figure 6 MIKE BASIN setup for East Rapti River Basin

It is possible to include each service provided by environmental flows as a water user (with 100% return flow) in the model set-up. The water demand time series would then be determined by the flow needed to maintain a service provision index (SPI) equal to 1. However, it is not always possible to specify a water demand time series. This is due to the fact that SPI curves may be based on flow classes that cannot be transferred into time series. For example, if a SPI curve is based on the flow of an annual flood event, this flow demand cannot be meaningfully represented by a time series. It is, therefore, recommended to post-process the water available for environmental flows in order to calculate SPI and evaluate scenarios.

In MIKE BASIN, it is possible to specify 'minimum flow requirements'. However, in the context of informing decision-making in IWRM, this is not an appropriate way of incorporating environmental flows into the model set-up as it does not allow for evaluation of various environmental flows scenarios and associated trade-offs.

Once the tool is set up, it can be used to explore and evaluate water allocation scenarios in a transparent manner. The tool can be used in different ways in the decision-making process, as will be discussed in the following section.

5.3 Decision-making processes

Several methods exist for evaluating scenarios and arriving at a decision. Such decision-making processes include the widely applied Cost-Benefit Analysis (CBA) and Multi-Criteria Analysis (MCA). For a thorough discussion of decision-making processes in relation to environmental issues, see Pearce et al. (2006). In the following, the tool described in this thesis will be placed in the general context of decision-making in IWRM.

In IWRM, there are three overriding criteria/guiding principles for decision-making (GWP, 2000):

- *Economic efficiency in water use*: Because of the increasing scarcity of water and financial resources, the finite and vulnerable nature of water as a resource, and the increasing demands upon it, water must be used with maximum possible efficiency;
- *Equity:* The basic rights for *all* people to have access to water of adequate quantity and quality for the sustenance of human well-being must be universally recognised;
- *Environmental and ecological sustainability:* The present use of the resource should be managed in a way that does not undermine the life-support system thereby compromising use by future generations.

The tool presented in the previous section can be used to evaluate scenarios and inform decision-making with respect to all three criteria. In MIKE BASIN, equity issues can be accommodated, while the SPI can take environmental and ecological sustainability into account. Finally, economic efficiency in water use can be explored using MS Excel, as all water uses, including environmental flows, are associated with economic values. In Figure 4 these three ways of evaluating scenarios are illustrated by the three arrows pointing to the decision support tool. As was mentioned in section 3.1, economic values will, in the case of perfect knowledge, take equity and sustainability into account. Therefore, in principle/theory all criteria could be merged into the economic criteria and thus be evaluated in a CBA. However, in a real world context, perfect knowledge does not exist, and it is, therefore, important to include all three criteria in decision-making processes in IWRM. This is best done by using a MCA. Alternatively, vital ecosystem services that are deemed important for political reasons may be accounted for in the CBA by assigning extremely high (infinite) values to such ecosystem services or by including them as constraints.

Notwithstanding the need to consider all three criteria, the focus of this thesis is on mainstreaming ecosystems and environmental flows into the economic rationales that, whether we like it or not, strongly influence decision-making. In the following some key aspects of this are addressed.

5.3.1 Optimal provision of ecosystem services

Figure 7 shows the relationship between ecosystem service provisions (ES) and the marginal costs (MC) and value (demand (D)) of service provision.

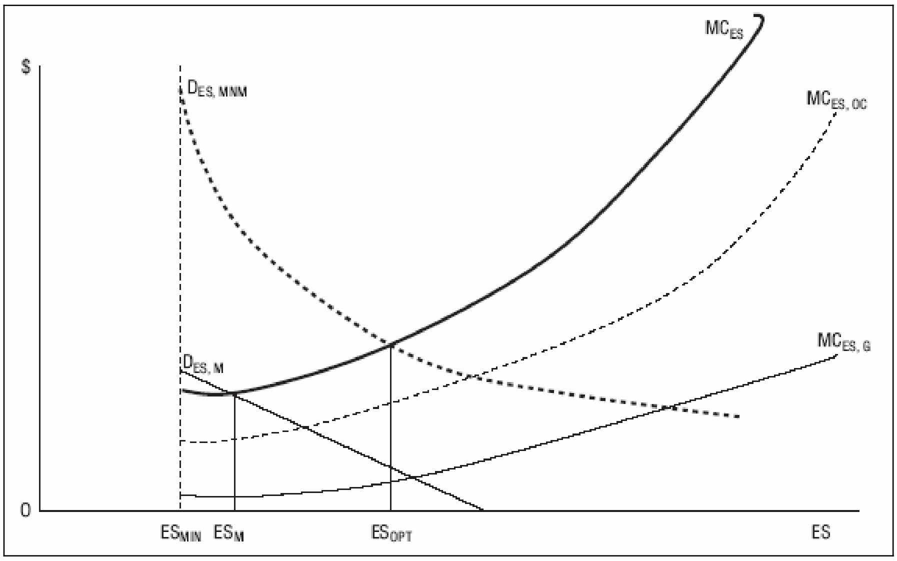

Figure 7 Stylised costs and values of ecosystem service provision. Source: Pearce et al. (2006).

$D_{ES,M}$ = demand curve for (marginal value of) marketed services

$D_{ES,NM}$ = demand curve for (marginal value of) non-marketed services (not shown)

$D_{ES,MNM}$ = total demand curve for (marginal value of) services ($D_{ES,M} + D_{ES,NM}$)

$MC_{ES,G}$ = marginal cost of service provision

$MC_{ES,OC}$ = marginal opportunity cost of service provision

$MC_{ES} = MC_{ES,G} + MC_{ES,OC}$

Figure 7 illustrates that from a utilitarian point of view, there is an optimal provision of ecosystem services (ES_{OPT}) when the marginal value of marketed and non-marketed services ($D_{ES,MNM}$) equals the total marginal cost (MC_{ES}). This optimal level of service provision is case specific and may be lower than the highest/pristine level. If only marketed services are considered ($D_{ES,M}$), this will give a lower economically optimal level of service provision (ES_M). Beyond a certain minimum service provision ES_{MIN} the very existence of humans is threatened, and some argue that the marginal value approaches infinity (Turner et al., 2003). However, others reason that the concept of economic value is meaningless below such a minimum provision (Pearce et al., 2006). The latter assertion is supported by the underlying assumptions of marginal change and substitutability as well as the practical/operational notion of budget constraint.

In IWRM, optimal environmental flows allocation is the flow at which the marginal value of service provision equals the marginal cost of providing that flow (incl. opportunity costs). While the marginal value of ecosystem services do differ between river basins, the marginal opportunity cost ($MC_{ES,OC}$) probably varies more than the other components in Figure 7. Therefore, opportunity cost is in fact the key determinant of optimal provision of ecosystem services.

5.3.2 Discounting

Discounting refers to the process of assigning a lower (in the case of a positive discount rate) weight to costs and benefits occurring in the future than to those occurring in the present. Discounting has a theoretical rationale in welfare economics and the discount rate is the sum of two fundamentally distinct components: the pure time preference rate and the growth rate of consumption per capita multiplied by the elasticity of the marginal utility of consumption (Turner et al., 1994). The former component arises as people inherently prefer the present over the future (particularly in the face of risks and uncertainties), while the latter follows from the diminishing marginal utility of consumption in a growing economy.

Discounting significantly influences the evaluation of scenarios. Using a constant positive rate has been termed the ‘tyranny of discounting’ as it discriminates against the future generations and thus is inconsistent with notions of intergenerational fairness and sustainable development (Turner et al., 1994). However, not discounting is the same as using a discount rate of 0, which also has unacceptable ethical implications (e.g., it discriminates against the present (poorest) generation; Pearce et al. (2006). A possible solution is to apply discount rates that decline with time. It has been observed that people actually have declining pure time preferences and there are several theoretical rationales to support declining discount rates (ibid.).

5.4 Stakeholder involvement - getting a negotiated response

The approach put forward in this thesis focuses on the end-results to people of providing environmental flows and sustaining ecosystem services. It is, therefore, important to involve the affected people, e.g. the stakeholders. The relevant stakeholders to involve depend entirely on the objectives of the environmental flows assessment and thus on the political issues addressed and prioritized. This, in turn, frames/scopes the assessment and defines the spatial, socio-economic and temporal scales to be considered (see section 3.3 and Paper C).

In relation to environmental flows assessment it is useful to distinguish between two main groups of stakeholders: (1) the above-mentioned stakeholders that are directly affected by ecosystem services provided by environmental flows, and (2) the stakeholders representing all other water uses in the river basin, e.g. irrigation, industry etc.

The two groups of stakeholders are involved in different parts of the process. The first group should be involved in the identification and valuation of important ecosystem services (see section 3.3). Both stakeholder groups should then be involved in evaluating the trade-offs between various water allocation scenarios and arriving at a negotiated solution. This solution will then determine the amount of water allocated for environmental flows and the resulting ecosystem condition and level of service provision. In Australia they have introduced the concept ‘Working Rivers’

(Whittington, 2002) and define a healthy working river as “a managed river in which there is a sustainable compromise, agreed to by the community, between the condition of the natural ecosystem and the level of human use”. The more work (e.g. hydropower, irrigation) a river is set to do, the less natural it becomes. In other words using the terminology in section 5.3.1: the higher the opportunity cost (see Figure 7 above) the lower the optimal ecosystem service provision. A compromise may be found between the level of work and the loss of naturalness, depending upon the values the community places on the river.

A crucial task in stakeholder involvement is to identify and reach all relevant stakeholder groups and ensure a participatory process. This requires stakeholders to see clear incentives to participate. Otherwise they are not likely to invest their precious time and energy in the process (Hermans et al., 2006). Further, a sense of ownership and responsibility will decrease the risk that the process stagnates. Experts, on the other hand, are required to facilitate the participatory process and act as brokers in the case of conflicts. Such facilitation and negotiation skills are as important as analytical skills and scientific knowledge. Stakeholder involvement is often a delicate balance between equal representation (empowerment of the poor) and practical manoeuvrability within existing decision-making and power structures.

Some of the more intangible services, for example carbon sequestration and biodiversity conservation, do not lend themselves easily to stakeholder assessment. Therefore, experts form an important stakeholder group that can speak on behalf of the ‘silent’ or ‘diffuse’ beneficiaries (‘expert participation’ as opposed to ‘expert consultation’).

A successful participatory approach not only ensures that stakeholder judgement/knowledge is incorporated into the valuation of ecosystems services. It also enables communication and learning among stakeholder groups (including experts). Furthermore, it establishes processes and builds capacity within the local civil society to participate in Integrated Water Resources Management (IWRM).

5.5 Dealing with uncertainties

The current knowledge of the links between environmental flows and ecosystem services is in many cases insufficient (Millennium Ecosystem Assessment, 2005). When subjected to changing flow conditions, ecosystems may exert non-linear and/or hysteretic behaviour. A change may cause cascading effects and lead to catastrophic and/or irreversible responses. On the other hand, some ecosystems may show strong resilience. It is, therefore, crucial to identify spatial/temporal thresholds, and extrapolations can only be used with great caution (Limburg et al., 2002; National Research Council, 2005). While increased understanding of ecosystem’s behaviour may reduce uncertainties, such understanding can only confirm the existence of non-linearity if it is present, it cannot prove its absence (ibid.).

Ecosystems may be resilient and able to cope with variability (e.g. droughts), but a lower provision of ecosystem services during these periods may be detrimental to human livelihoods. This depends on the resilience and coping strategies of the affected population and introduces a further dimension of uncertainty.

Decision-making in the presence of such large uncertainties should proceed with caution. This implies the adoption of the precautionary principle, safe minimum standards, strong sustainability constraints or other safeguarding principles (Pearce, 2006). The extent to which these principles are adhered to will inevitably vary with the risk profile of the decision-makers (the societies).

As uncertainties can be reduced over time through passive and/or active learning, it has been argued that the value of postponing a decision until more information is available must be considered (National Research Council, 2005; Pearce et al., 2006). Such value is referred to 'quasi option value' (Pearce et al., 2006). While delaying a decision may be justified in some cases, it is counterproductive to the urgent need to demonstrate the roles and values of environmental flows and ecosystem services for human well-being. One could argue that the 'quasi option value' must be compared to the cost of no-action.

Under conditions of uncertainty, irreversibility, and learning, there should be a clear preference for adaptive management. Adaptive management provides a mechanism for learning systematically about the links between flows, services and value. When establishing these links in the face of lacking knowledge and significant uncertainty, it is necessary to make expert judgements. This suggests that in environmental flows assessments there is a strong case to include peer/stakeholder reviews providing inputs and incorporating some measure of quality assurance.

6 Case study: East Rapti River Basin, Nepal

6.1 Introduction

This case study presents a rapid application of the SPI approach to assess environmental flows in an IWRM context. Due to data and time constraints, many assumptions and expert judgements are made, and issues related to participation *per se* are not addressed. The following sections include description of the study area, quantification of the water uses and estimation of the value of water uses. Details of the SPI calculations are given in Paper D. The last section then provides examples of scenarios that may define a future decision space.

6.2 Physical Characteristics

6.2.1 Location, topography and land use

East Rapti River is located in Nepal, southwest of Kathmandu and is a major tributary to the Narayani River (Figure 8). The catchment area of East Rapti River is approximately 3100 km^2. The north-eastern part of the basin is mountainous with altitudes of more than 2000 m a.s.l., while the south-western floodplains lie at altitudes below 400 m a.s.l.. These floodplains, also known as Inner Terai, comprise some of Nepal's most fertile agricultural land as well as the Royal Chitwan National Park, which is a World Heritage Site. The Royal Chitwan National Park covers 25 % of the total basin area. Agricultural land and secondary forest covers approximately 30% and 40%, respectively, while the remaining 5% of the basin is urban development and infrastructure (Shilpakar, 2003).

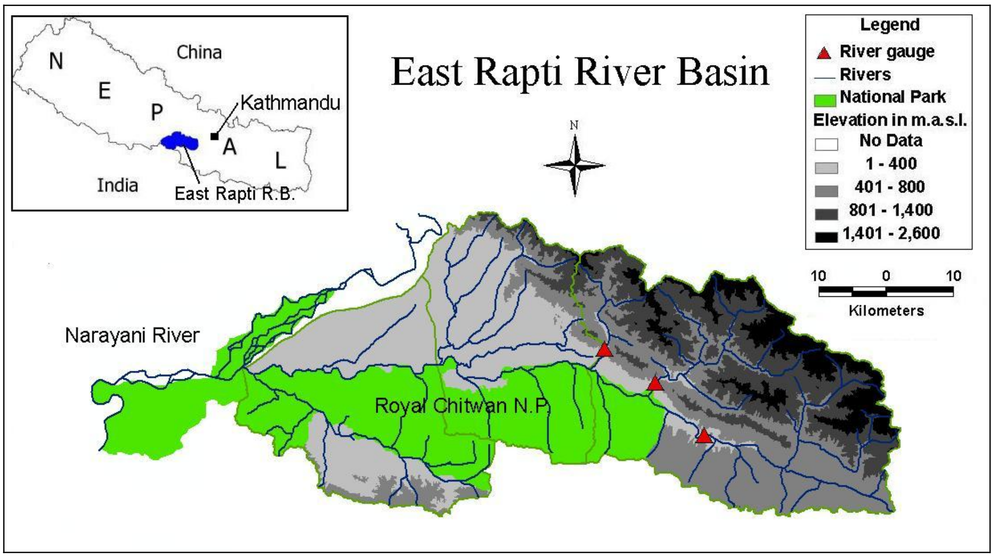

Figure 8 East Rapti River Basin, showing elevation, major tributaries, the location of existing flow stations and the area of Royal Chitwan National Park.

6.2.2 Climate

The basin has a subtropical to tropical climate with relative humidity varying between 50% and 90% and an average annual rainfall of approximately 2000 mm of which about 80% falls in the summer monsoon from May to October (see Figure 9 below). Average annual potential evaporation is 1300 mm, while mean daily temperatures range from 5 to 30 °C.

6.2.3 East Rapti River

The river is not regulated, but is receiving water from Kulekhani Hydropower Plant in the adjacent Bagmati River Basin and from the Narayani Lift Irrigation System. There are three relatively reliable and well calibrated flow gauges in the basin (see Figure 8). Daily flow data is available from 1963-1995 and average runoff is shown in Figure 9.

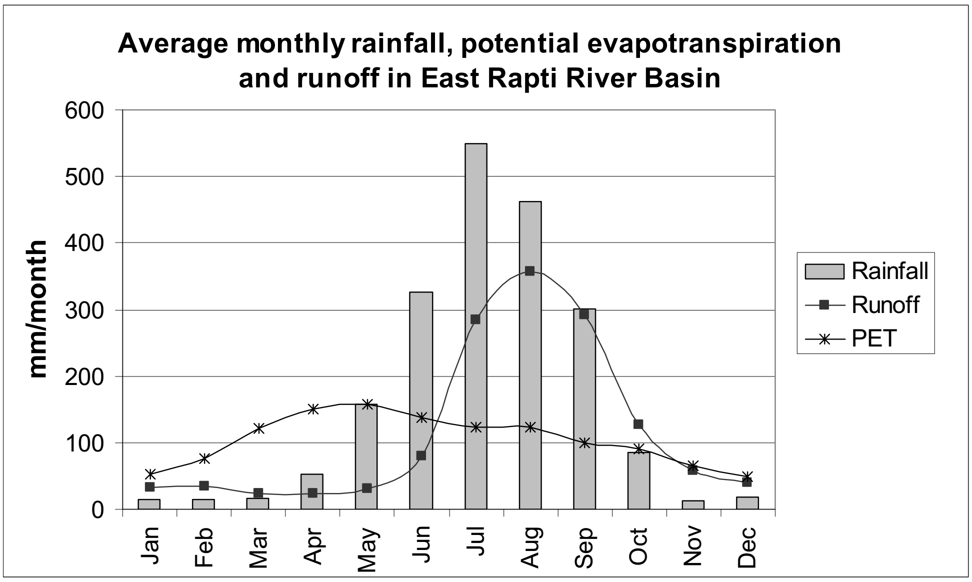

Figure 9 Average monthly rainfall, potential evapotranspiration (PET) and runoff in East Rapti River Basin (1963-1995). Data source: Department of Hydrology and Meteorology, Kathmandu, Nepal.

6.2.4 Main water related problems

- Water scarcity experienced during dry-season:
  → more groundwater abstraction, rainwater harvesting, changing of cropping pattern
  → inter and intra sectoral conflicts between water users/uses (e.g. tourist industry/domestic/irrigation)
- Decline in fish population due to changed flow regime, water quality (industrial effluents, poison, gelatine, and explosives used for fishing), lack of 'moss':
  → marginalization of fishermen
- Sand mining causes river bed to change position and river banks to destabilize
  → flooding (as river bed in some places is elevated), intakes above water level, erosion

6.3 Ecosystem services sustained by environmental flows

Table 6 is a checklist of all the possible services sustained by environmental flows. Services highlighted in light grey are present in East Rapti River Basin, and services highlighted in dark grey are considered the most important in the Basin. The selection of the most important services was based on expert judgement during a field visit.

Table 6 A checklist of services sustained by environmental flows (Paper A).

Services category	Service provided
Production	Water for people - subsistence/rural
	Fish
	Fertile land for flood-recession agriculture and grazing
	Wildlife for hunting
	Vegetables and fruits
	Fibre/organic raw material for building/firewood
	Medicine plants
	Inorganic raw material for construction and industry (gravel, sand, clay)
Regulation	Chemical water quality control (purification capacity)
	Physical water quality control
	Flood mitigation
	Groundwater replenishment (low flow maintenance)
	Health
	Pest control
	Erosion control (riverbed/bank dynamics)
	Prevention of saltwater intrusion (salinity control)
	Prevention of acid sulphate soils development
	Carbon "trapping" (sequestration)
	Microclimate stabilization
Information	**Recreation and tourism opportunities**
	Biodiversity conservation
	Cultural/religious/historical/symbolic activities
Life support	The prior existence of healthy ecosystems

Table 7 shows details of the important services. Three of these services, namely fish production, recreation and tourism, and biodiversity conservation will be subjected to further analysis.

Table 7 Most important services sustained by environmental flows in East Rapti River Basin, their type of value and appropriate valuation method

Service provided	Key flow related function	Type of Value	Valuation method
Fish production	Habitat availability and connectivity, food supply	Direct use	Market price
Groundwater replenishment (low flow maintenance)	Groundwater (aquifer) replenishment	Indirect use	Replacement cost/mitigative expenditure or damage cost avoided
Erosion control (riverbank/bed dynamics)	Healthy riparian vegetation, erosion, transportation and deposition of sediments	Indirect use	Replacement cost/mitigative expenditure or damage cost avoided
Recreation and tourism opportunities	Presence of wildlife, aesthetic significance, good water quality	Direct and indirect use	Travel cost, revenue from tourists
Biodiversity conservation	Sustaining ecosystem integrity (habitat diversity and connectivity)	Option, bequest, existence	Benefit transfer

In order to quantify environmental flows in East Rapti River Basin following the methodology proposed by Paper D the links between flows, services and values must be established. The main links to be dealt with are:

- The links between flow and the recreational value of Royal Chitwan NP
- The links between flow and fish production
- The links between flow and biodiversity conservation

Table 8 lists the information required to establish the required links.

Table 8 Information/data requirements.

Service provided	Info needed to link to flow	Info needed to link to value
Fish	Suitability curves for most important species	Present catch Market price
Recreation and tourism	Suitability curves for most important wildlife and most significant aesthetic locations	Revenue/income from tourism (park, hotels, local population, guides, concessionaires)
Biodiversity conservation	Suitability curves for most important species	Benefit transfer: finding existing valuation study with similar context

6.4 Water uses

6.4.1 Agriculture

One of the main water users in the basin is irrigated agriculture. Water for the gravity-fed irrigation systems is diverted directly from East Rapti River. The cropping intensity is about 180% and the crop coverage is given in Table 9 which also shows reference crop evapotranspiration, effective rainfall and crop coefficients.

Table 9 Reference crop evapotranspiration ET_0 (mm) effective rainfall (mm), crop coefficients K_c and crop coverage Source: FAO (1986), IWMI (2000), Shilpakar (2003).

Month	ET_0	Effective rainfall	Wheat	Paddy (spring)	Paddy (main)	Maize (winter)	Maize (summer)	Oil-seed	Potato	Pulses	Vege-tables
Jan	53	1	0.85			1.04		1.00	0.33	0.70	
Feb	75	1	1.06			0.83		1.09	0.67	0.15	
Mar	121	1	0.80	0.57		0.23	0.16	0.73	1.00		
Apr	150	23	0.06	1.14			0.48	0.07	1.00		
Maj	158	103		1.24			0.98		0.78		
Jun	138	245		1.24	0.46		1.04		0.33		
Jul	124	412		0.71	0.97		0.54				
Aug	124	347			1.07						0.20
Sep	99	226			1.05						0.60
Okt	90	43			0.97	0.14				0.28	0.93
Nov	66	1			0.44	0.47		0.12		0.87	0.93
Dec	50	2	0.25			0.91		0.46		1.10	0.33
Crop coverage (% of agricultural area)			17	7	51	40	16	30	3	10	5

Based on the information in Table 9, crop water demands have been calculated using FAO guidelines (FAO, 1986):

$$ET_c = K_c ET_0 \qquad [1]$$

where

ET_c = crop evapotranspiration under standard conditions (no water stress)

ET_0 = reference crop evapotranspiration

K_c = crop coefficient

Crop water demands in mm/months are shown in Figure 10.

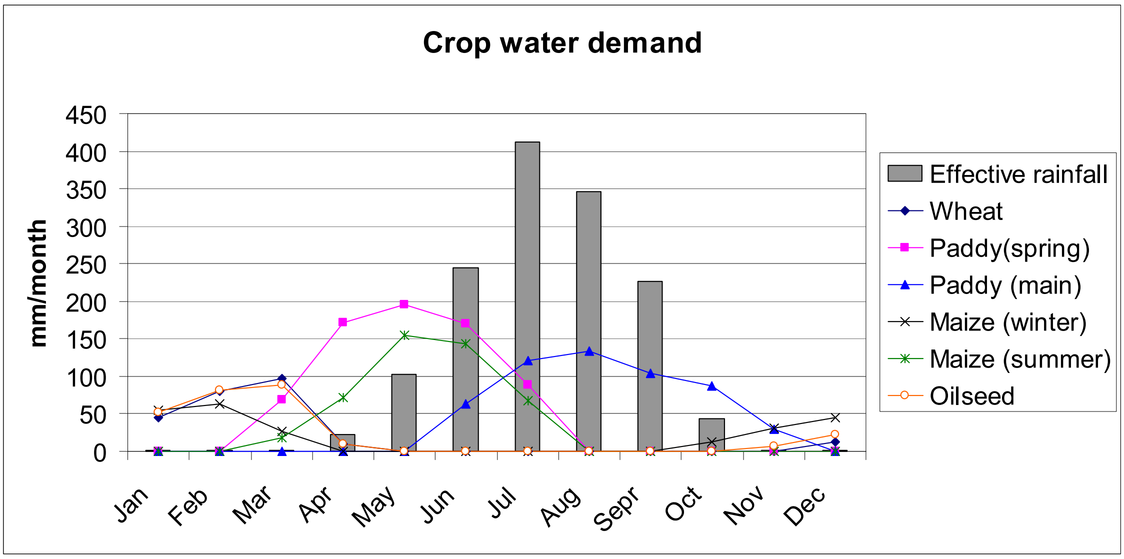

Figure 10 Effective rainfall and water demand of most important crops.

The total irrigation water demand depends on the size of the agricultural area and the effective rainfall. In the base line scenario the agricultural area is estimated at 480 km^2. The irrigation water demand for this scenario is given in Table 10. Note that this demand includes both consumptive water use by the crops as well as water lost in the distribution system - including evaporation from canals (consumptive water use) and leakage through the canals (non-consumptive use). It must be noted that while such water 'losses' may be considered pure losses from an agricultural point of view, they are not lost from the river basin and may be beneficial in other respects. This includes climate regulation and groundwater replenishment.

Table 10 Irrigation water demand in East Rapti River Basin

	Irrigation demand (m^3/s)									Total irrigation demand	
	Wheat	Paddy (spring)	Paddy (main)	Maize (winter)	Maize (sum.)	Oil-seed	Potato	Pulses	Vege-tables	m^3/s	10^6 m^3/year
Jan	1.7			5.0		3.6	0.1	0.8		11.3	29
Feb	3.1			5.7		5.6	0.3	0.2		15.1	39
Mar	3.8	1.1		2.4	0.7	6.1	0.8			14.9	39
Apr		2.4			1.8		0.9			5.1	13
May		1.5			1.9		0.1			3.6	9
Jun										0.0	0
Jul										0.0	0
Aug										0.0	0
Sepr										0.0	0
Oct			5.2						0.5	5.2	13
Nov			3.3	2.8		0.5		1.3	0.7	8.5	22
Dec	0.4			4.0		1.4		1.2	0.2	7.2	19
Total	9.0	5.0	8.5	19.9	4.4	17.2	2.3	3.6	1.3	71.0	184

6.4.2 Industrial and domestic water use

The main industrial area comprises more than 130 industries. The types of industries and their annual water demand in year 2000 are shown in Table 11.

Table 11 Industrial water demand in East Rapti River Basin

Type of industry	Brew-ery	Con-crete	Dairy	Food	Leader	Chemi-cals	Plas-tic	Tex-tile	Gee	Wood	Other	Total
Water demand (10^6 m^3/year)	0.29	0.01	0.02	0.09	0.03	0.07	0.01	0.03	0.03	0.01	0.08	0.69

The total population of the basin in 2001 was approximately 860,000 of which about 80% are employed in the agricultural sector (Central Bureau of Statistics, 2004). Only 22% of the population live in the three main urban centers (Hetauda, Narayanghat-Bharatpur and Ratnanagar), but the area is undergoing rapid urbanization. Table 12 shows the estimated domestic water demand based on the assumption that rural and urban populations need 45 and 60 l/c/d, respectively.

Table 12 Domestic water demand in East Rapti River Basin

	Total population in 2001	Water supply (l/c/d)	Estimated water demand (10^6 m^3/year)
Rural population	669056	45	10
Urban population	195596	60	4.3
Total	864652		14.3

6.4.3 Environmental Flows

Smakhtin et al. (2006) applied two different desktop methods to estimate the environmental flows requirement in East Rapti River - the Tennant Method and Range of Variability Approach (RVA). The results of these assessments are shown in Table 13 and compared to two existing recommendations by the Ministry of Water Resources in Nepal.

Table 13 Results of 4 different Environmental Flow assessments in East Rapti River. MAR = mean annual runoff. Based on Smakhtin et al. (2006).

Method	Resulting Environmental Flow Requirement	Water demand (10^6 m^3/year)
Ministry of Water Resources, 2001 (10 % of min. monthly avg. flow)	2 m^3/s (1.6% of MAR)	63
Ensure navigation (Depth = 1 m, width = 50 m, velocity = 0.3 m/s)	15 m^3/s (12% of MAR)	470
Tennant Method - poor ecological status	10% of MAR	260
Tennant Method - good ecological status	60% of MAR	1600
Range of Variability Approach (RVA)	56% of MAR	1500

All the applied methods have considerable drawbacks and limitations. The most critical issue is that all methods return an environmental flows requirement that is an arbitrary and constant fraction of the natural flow regime. Mimicking the natural flow regime but at a lower level may not be the most ecologically sound/optimal way of defining an environmental flows requirement, and may even result in a 'waste of water' from a socio-economic point of view. The Service Provision Index (SPI) approach circumvents these problems by defining the environmental flows requirement as a negotiated trade-off between water uses.

6.4.4 Summary of water uses

Table 14 Summary of water uses in East Rapti River Basin. [1]Environmental Flows requirement based on existing methods. [2] Environmental Flows requirement based on the Service Provision Index (SPI).

Water use	Annual water demand (10^6 m^3/year)
Irrigated agriculture	180
Industry	0.7
Domestic	14
Environmental flows[1]	60 – 1600
Environmental flows[2]	0 – 2600
Natural flow	2600

Industrial and domestic water uses will not be subjected to further analysis as they are negligible compared to the agricultural water use and environmental flows.

6.5 Value of water uses

In this case study, a financial economic analysis is undertaken. The spatial scale is thus defined by the value added directly to the river basin by services sustained by environmental flows. Travel cost, therefore, is not included in the valuation of recreation/tourism. Regarding the temporal scale, it is assumed that all service provisions have the same discount rate or that all the benefits occur immediately (in year 0). Therefore, discounting is not applied.

6.5.1 Agriculture

The value of water used for agricultural production is estimated based on the market price of the crops produced. There are two major assumptions to be made when using this method: (1) Water is the limiting factor for production - e.g. that water availability is directly related to crop yield in a given area. (2) Crop production is the only benefit of irrigation water.

Crop yield may then be calculated using FAO guidelines (FAO, 1998):

$$Y_a = -Y_m K_y (1 - \frac{ET_{cadj}}{ET_c}) + Y_m \qquad [2]$$

where:

Y_a = Actual crop yield

Y_m = Maximum expected crop yield (no water stress)

K_y = Yield response factor

ET_{cadj} = Adjusted actual crop evapotranspiration

ET_c = Crop evapotranspiration for standard conditions (no water stress)

ET_{cadj} is calculated using:

$$ET_{cadj} = K_s K_c ET_o \qquad [3]$$

where:

K_s = Water stress coefficient

K_c = Crop coefficient

ET_0 = Reference crop evapotranspiration (mm/d)

The water stress coefficient is calculated using:

$$K_s = \frac{TAW - D_r}{(1-p)TAW} \qquad [4]$$

where:

TAW = Total available soil water in root zone (mm)

D_r = root zone depletion (mm)

p = fraction of TAW that a crop can extract without suffering water stress

TAW is calculated as follows:

$$TAW = 1000(\theta_{FC} - \theta_{WP})Z_r \quad [5]$$

where:

Θ_{FC} = water content at field capacity (m^3/m^3)

Θ_{WP} = water content at wilting point (m^3/m^3)

Z_r = rooting depth (m)

The root zone depletion for a given time step is calculated based on the water demand deficit of the preceding time step. Table 15 summarises the parameters used to calculate crop yield.

Table 15 Parameters used to calculate water stressed yield. Maximum expected crop yield (Y_m) is based on FAOSTAT (2005), while yield response factor (K_y), fraction of soil water that a crop can extract without suffering water stress (p), rooting depth (Z_r), water content at field capacity (Θ_{FC}) and at wilting point (Θ_{WP}) are found in FAO (1998).

Crop	Y_m (t/ha)	K_y	P	Z_r (m)	Θ_{WP}-Θ_{FC} (m^3/m^3)
Wheat	1.9	1.05	0.67	1.5	0.15
Paddy (spring)	2.7	1.00	0.56	0.75	0.15
Paddy (main)	2.7	1.00	0.59	0.75	0.15
Maize (winter)	1.9	1.25	0.66	1	0.15
Maize (summer)	1.9	1.25	0.66	1	0.15
Oilseed	0.5	1.00	0.72	1.25	0.15
Potato	9.9	1.10	0.46	0.5	0.15
Pulses	0.7	1.00	0.61	0.7	0.15
Vegetables	10.2	1.00	0.56	0.7	0.15

The value of crop production in East Rapti River Basin is then calculated by using market prices. For the base line scenario, the value of agricultural production is given in Table 16. All prices are adjusted to 1998 levels. Production cost includes inputs (seeds and fertilizers) (OBS: Irrigation water is provided free of charge), labour (humans and animals) and machinery.

Table 16 Net annual revenue from agricultural production in East Rapti River Basin. [1]Market prices and production costs are based on Asian Development Bank (2002).

Crop	Cropped area (km^2)	Yield (t/ha)	Market price[1] (NRs/t)	Gross revenue (NRs/ha)	Production cost[1] (NRs/ha)	Net revenue (NRs/ha)	Net annual revenue (mill.NRs)
Wheat	102	1.9	8000	15200	15000	200	2
Paddy (spring)	42	2.7	8500	22145	20000	2145	9
Paddy (main)	306	2.7	8500	22145	20000	2145	66
Maize (winter)	240	1.9	7300	13240	13000	240	6
Maize (summer)	96	1.9	7300	13240	13000	240	2
Oilseed	180	0.5	21000	10500	9000	1500	27
Potato	18	9.6	5500	54382	40000	14382	26
Pulses	60	0.7	20000	13954	7000	6954	42
Vegetables	30	10.2	5500	56057	40000	16057	48
Total							230

6.5.2 Environmental flows

6.5.2.1 Fish production

Only villages depending primarily on fisheries for income generation were included in the field survey and used to estimate fish production. The actual fish catch may be somewhat higher, as many farmers also rely on fishing as their main source of proteins.

In the study area, three fishing villages, with a total population of 500, were identified. In each village 4-5 fishermen were responsible for most of the fish catch. They reported an average daily catch of 4-5 kg/fisherman during the main season, and 30-50 kg/fisherman during the rest of the year. The market price of fish varies between 120-150 NRs/kg, depending on size and species. Mainly catfish, snakefish and prawnfish are caught. The fishermen all mentioned that their catch had declined in the past five years. The reason for this decline they believed to be caused by increased sedimentation in the riverbed and reduced water quality.

Using the figures obtained during the field visit, the total value of the current fish catch is given in Table 17.

Table 17 Baseline value of fish production in East Rapti River.

Gross value of fish catch	Average	Range
mill NRs/year	1.0	0.6-1.5
mill US$/year	0.015	0.01- 0.02

To check the value, a calculated average annual per capita income was compared to the official Nepali survey on income in 2003/04. This survey revealed that the poorest 20% of the population had an average income of 4000 NRs/c/year. The income in the visited fishing villages was on average 2000 NRs/c/year with a range of 1400-4000 NRs/c/year. These fishing villages are considered the poorest in the study area. Thus, the figures reported in the current study seem realistic.

It is worth noting that while the economic value of fish production is insignificant compared to that of recreation and tourism (see below), fish production supports the poorest people in the basin and is the only source of proteins to these people. In this case, the value of fish production may actually approach the cost of changing livelihood strategy. Thus, using the market price method is not appropriate in the context of livelihood subsistence and poverty. Is does, however, provide a minimum value.

6.5.2.2 Recreation/tourism

The value of recreation/tourism was estimated using information on the number of tourists visiting the area as well as the income generation in the two main tourist centers, Sauraha and Baghmara.

Net-revenue generated by visitors to Royal Chitwan National Park:

Table 18 Number of visitors to Royal Chitwan National Park and the net annual revenue. Source: Ministry of Forests and Soil Conservation (2004).

year	Total no. of visitors	Net annual revenue (mill. NRs)
1997-98	104046	50
1998-99	105884	55
1999-00	117000	52
2000-01	132922	75
2001-02	83073	40
2002-03	n.a.	31
2003-04	56389	41
Mean	100000	50

Net income generation in Sauraha:
Approximately 1000 persons are directly involved in tourism (guides, boat drivers, hotel/restaurant/shop staff) and approximately 3000 persons indirectly benefit from the activities related to tourism. The total net income generation in 2005 is estimated at 100 mill. NRs.

Net income generation in Baghmara:
Approximately 150 persons are directly involved in tourism (guides, boat drivers, hotel/restaurant/shop staff) and approximately 5000 persons indirectly benefit from the activities related to tourism. The total net income generation in 2005 is estimated at 3 mill. NRs.

Total value of tourism
Table 19 summarises the value added by tourism in East Rapti River Basin.

Table 19 Total value added by tourism in East Rapti River Basin Source: Ministry of Forests and Soil Conservation (2004).

	Mill. NRs	Mill. $
Park revenue	50	0.7
Sauraha	100	1.4
Baghmara	3	0.04
Total		2.1

6.5.2.3 Biodiversity conservation

From Paper C, the annual value of biodiversity conservation in developing countries is found to be in the range of US$ 1 to US$ 30 per ha. Using the per hectare unit assumes that biodiversity conservation is a function of the size of the conservation area, which is a reasonable assumption. Given that the Royal Chitwan National Park has a size of

0.0932 mill. ha, simple benefit transfer suggests a value of biodiversity conservation in East Rapti River Basin of 0.1-3 mill. $/year, with a mean of 1.5 mill. $/year.

6.5.3 Summary of values

Table 20 Most important services sustained by environmental flows in East Rapti River Basin, their type of value, appropriate valuation method, and their estimated net value. Values in brackets indicate the range.

Service provided	Type of Value	Valuation method	Net Value (mill. US$/year)
Fish production	Direct use	Market price	0.015 (0.01-0.02)
Recreation and tourism opportunities	Direct use	Revenue from tourists	2.1 (1.9-2.5)
Biodiversity conservation	Option, bequest, existence	Benefit transfer	1.5 (0.1-3.0)
Total value added by ecosystems services sustained by environmental flows			3.6 (2.0-5.5)

6.6 Scenario evaluation and discussion

There are plans of increasing the irrigated area in East Rapti River Basin, from the current 480 km^2 to 600 km^2 (scenario 1) or even 800 km^2 (scenario 2). This would reduce the water available for environmental flows and thus cause a reduction in service provision. In order to investigate the trade-offs and provide information to decision-makers, a decision support system was set up in MIKE BASIN and MS Excel as described in Chapter 5 and Paper D. Table 21 summarizes the results of a scenario analysis.

Table 21 Summary of values (and range of values) associated with environmental flows and irrigated agriculture for three different water allocation scenarios in East Rapti River Basin. In the base-line scenario, the irrigated area is 480 km^2, whereas the irrigated area is 600 km^2 and 800 km^2 for future scenario 1 and 2, respectively.

Service provided	Net Value (mill. US$/year)		
	Base-line scenario	Future scenario 1	Future scenario 2
Fish production	0.015 (0.01-0.02)	0.013 (0.01-0.02)	0.012 (0.01-0.014)
Recreation and tourism opportunities	2.1 (1.9-2.5)	1.9 (1.4-2.0)	1.8 (1.3-1.9)
Biodiversity conservation	1.5 (0.1-3.0)	1.0 (0.08-2.3)	0.9 (0.06-1.8)
Total value added by ecosystems services sustained by environmental flows	3.6 (2.0-5.5)	2.9 (1.5-4.3)	2.7 (1.4-3.7)
Value added by irrigated agriculture	3.1 (1.2-4.8)	3.4 (1.4-5.6)	2.6 (1.4-5.3)
Total value added	6.7 (3.2-10.3)	6.3 (2.9-9.9)	5.3 (2.8-9.0)

Ecosystem services, sustained by environmental flows, are not presently included in water management decisions in East Rapti River Basin. This study shows, however, that the present annual value of such ecosystem services, approximately 3.6 million US$, is at the same order of magnitude as the value of the agricultural production, approximately 3.1 million US$.

The study also shows that augmenting water withdrawals for agriculture will reduce environmental flows and associated values. In scenario 2, the value of agricultural production is lower than in the other scenarios. This is due to the fact that there is not enough water during the dry season. Thus, expanding the irrigated area to 800 km^2 without additional developments such as dam construction is not a feasible development option in East Rapti River Basin.

The figures in Table 21 show a considerable range and additional research is needed to get more accurate results. In particular the value of 'biodiversity conservation' should be further investigated. If this value is omitted from the analysis is does not, however, affect ranking of the scenarios. The value of tourism is sensitive to the number of tourists visiting the area. However, in order for scenario 1 to break even with (rank as high as) the base-line scenario, the number of tourists must increase by 50%. As the economic value of fish production is insignificant, the ranking of scenarios is not sensitive to changes in the market price of fish. The ranking of scenarios is very sensitive to agricultural yield and market prices. If yield or market price of the main crop (paddy) increases by only 5% in scenario 1, this scenario will break even with the base line scenario. Therefore, further investigations are needed.

Nevertheless, the case study illustrates that the SPI approach is useful for including environmental flows into scenario analysis. It also shows that it is crucial to include ecological aspects into river basin management, as ecosystem services may support livelihoods of the poorest people. The case study highlights the shortcomings of using market price methods for estimating the value of such services that supports the livelihoods of people. If poverty reduction is on the agenda in a river basin, using the SPI index could serve as a better means of comparison between environmental flows scenarios than economic value.

While many uncertainties and shortcomings remain, using SPI and economic valuation of environmental flows seems to be a promising way of incorporating environmental flows into decision-making in IWRM.

7 Summary of papers and Network material

Paper A
Korsgaard, L., Jønch-Clausen, T., Rosbjerg, D. & Schou, J.S. (2005): Quantification of environmental flows in integrated water resources management. In: Brebbia, C.A. & Antunes do Carmo, J.S. (eds.): River Basin Management III, WIT Press, Boston. 141-150.

This paper examines the potentials for quantifying environmental flows in the context of Integrated Water Resources Management (IWRM). The paper introduces relevant concepts, definitions and approaches and summarises the findings of literature reviews of environmental flow assessment methodologies and economic valuation methods. It argues that none of the existing environmental flows assessment methodologies are readily applicable for IWRM and recommends the development of a holistic desktop method. A checklist, linking ecosystem functions, environmental services, types of value, and relevant valuation methods is presented and an operational tool, consisting of three components (MIKE Basin, an EFA, and MS Excel) is conceptualised. The paper concludes that ultimately it is the responsibility of decision-makers to select the water allocation scenario to be adopted, and thus to quantify environmental flows in IWRM.

Paper B
Korsgaard, L., Jønch-Clausen, T., Rosbjerg, D. & Schou, J.S. (in press): Using economic valuation of environmental flows to integrate ecological aspects into water management. In: Proceedings of the 3rd International Symposium on Integrated Water Resources Management, 26-28 September 2006, Bochum, Germany.

This paper addresses the potential for using economic valuation of environmental flows to incorporate ecosystem services into decision-making in Integrated Water Resources Management (IWRM). A water allocation decision support tool is presented and tested in East Rapti River Basin, Nepal. In East Rapti River Basin the main ecosystem services supported by environmental flows are biodiversity conservation, recreation and tourism opportunities, fish production, and sediment flushing. These services are valued using a combination of market price methods and cost-based methods. Thus, socio-economic and environmental implication of various water allocation scenarios can be evaluated. Preliminary findings suggest that water scarcity is a problem in East Rapti River Basin. The main trade-offs are between water for irrigation and water for environmental flows. Ecosystem services, sustained by environmental flows, are not presently included in water management decisions. The study shows, however, that the present annual value of such ecosystem services, approximately 3.6 million US$, is at the same order of magnitude as the value of the agricultural production, approximately 3.1 million US$. Furthermore, ecosystem services support livelihoods of the poorest people in the basin. Hence, it is crucial to include ecological aspects into water management. Economic valuation of environmental flows seems a promising way forward.

Paper C

Korsgaard, L. & Schou, J.S. (submitted): Economic valuation of aquatic ecosystem services in developing countries. Submitted to *Ecological Economics.*

This paper provides a critical review of recent literature on economic valuation of aquatic ecosystem services in developing countries and gives an overview of the state of the art and the main challenges. It finds that 'market price' is the most widespread method used for valuating marketed ecosystem services in developing countries. However, market distortions, limited access to markets, and subsistence use often violate the underlying assumptions of marginality and substitutability. 'Cost based' and 'revealed preference' methods are frequently used when ecosystem services are non-marketed. These methods are problematic when addressing subsidence use and/or people with a very restrictive budget constraint. Four main challenges for valuation of ecosystems services are identified: (1) acknowledging the assumptions of marginality and substitutability, (2) using 'total' economic value, (3) defining spatial, socio-economic and temporal scale, and (4) dealing with uncertainty. If these challenges are not well appreciated, the valuation study may be misleading or meaningless - regardless of the method chosen. This should not lead to rejection of economic valuation of ecosystems, nor should it render scientists paralysed or tempted to convey a false sense of precision. Instead, it should encourage careful and explicit attention to the caveats of economic valuation of ecosystem services. Such caveats are seldom explicitly accounted for in the literature. A review of 27 existing valuation studies reveals a considerable range of estimated total economic value of aquatic ecosystem services in developing countries, from 30 to 3000 US$/ha/year or from 10 to 230 US$/capita/year. The paper concludes that economic valuation is vital for bringing ecosystems to decision-making agendas in developing countries and that great efforts must be made to bridge the gap between scientists/academics and decision-makers/practitioners.

Paper D

Korsgaard, L., Jønch-Clausen, T., Rosbjerg, D. & Schou, J.S. (submitted): A service and value based approach to estimating Environmental Flows in IWRM. Submitted to *International Journal of River Basin Management.*

This paper presents the concepts and methodology of the new environmental flows assessment approach, the Service Provision Index (SPI). The paper further discusses the main advantages and disadvantages of the SPI approach in comparison with other existing environmental flows assessment methods. The main advantage of the SPI approach is that it explicitly links environmental flows to (socio)-economic values by deliberately focusing on ecosystem services. This is a novel contribution to the existing field of environmental flows assessment methodologies. Furthermore, the SPI approach is pragmatic and operational even in data-scarce applications. The main disadvantage is the fact that in such data-scarce applications, the links between flows and services are assumed rather than directly assessed. This is, however, also the case for existing rapid desk-top methods (e.g. hydrological methods). Compared to such methods, the SPI approach has the advantage that it can be refined, should more information become available. In summary, the SPI approach is a flexible, transparent and relatively rapid tool for incorporating ecosystems and environmental flows into the evaluation of water allocation scenarios, negotiations of trade-offs and decision-making in IWRM.

Material produced for the Global Environmental Flows Network

A part of this Ph.D. project has been dedicated to the establishment of a Global Environmental Flows Network.

The network aims at making the Environmental Flows concept accessible to all groups of stakeholders: river basin managers, policy-makers, NGOs, governmental and inter-governmental agencies and to a wider public. The network will provide access to Environmental Flows tools and knowledge and will act as an open portal for anyone interested in Environmental Flows, whether it is an interest in most basic concepts or specific technical questions. In other words, the network will serve as a central reference point where people can readily access or share all Environmental Flows related information.

The materials produced in relation to the Network include a concept note, a seminar programme for the event at World Water Week in Stockholm and a special issue of the environmental flows Newsletter. The material is produced in collaboration with Katharine Cross (The World Conservation Union, IUCN), Vladimir Smakhtin (International Water Management Institute, IWMI), Mike Acreman (Centre for Ecology and Hydrology, CEH), Karen Meijer (Delft Hydrualics), Karin Krchnak (The Nature Conservancy, TNC) and Michael Moore (Stockholm International Water Institute, SIWI).

8 Summary and Conclusions

This Ph.D. project has developed an operational tool for quantifying environmental flows in the context of Integrated Water Resources Management (IWRM). In this context, the environmental flows requirement is a negotiated trade-off between water uses. The trade-offs involved are inherently case-specific. So are the preferences and policies of decision-makers. In some river basins, for example, irrigated food production is vital and a low environmental flows requirement (and thus a low level of ecosystem service provision) is accepted. In other river basins, high environmental flows requirements are set in order to maintain valuable ecosystem services. It is all a matter of prioritizing the water uses and the associated trade-offs.

While several holistic and interactive environmental flows assessment methods have been developed, none of them explicitly links environmental flows to ecosystem services. Consequently, such methods cannot readily deliver inputs to economic valuation studies. Furthermore, existing holistic environmental flows assessment methods are very resource (time, money, data) demanding. This is a major constraint for undertaking environmental flows assessments - in particularly in developing countries. There is a need for developing a holistic desktop environmental flows assessment method that pays due attention to the ecosystem services provided to people.

A checklist of such ecosystem services related to environmental flows is presented in the thesis. The checklist also shows relevant economic valuation methods for each ecosystem services and gives the ranges of economic values reported in recent literature. The estimated total economic value of aquatic ecosystem services in developing countries ranges from 30 to 3000 US$/ha/year or from 10 to 230 US$/capita/year. '*Market price*' is the most widespread method used for valuating marketed ecosystem services, also in developing countries. '*Travel cost*' is often applied to estimate recreational values, while '*Stated preference*' methods are the preferred methods for valuating non-marketed services. However, preferences of wealthy people may get a higher weight than that of poor people and subsistence use may not be accounted for at all. While '*Cost based*' methods have been heavily criticised, they are widely used to estimate indirect use values. '*Benefit transfer*' is an easy desktop method, but it may give poor results if contexts differ.

Existing valuation methods have their drawbacks, but they offer adequate opportunities for raising awareness about the roles and values of ecosystem services for human well-being and thus for assessing the diverse suite of values associated with environmental flows allocations. Consequently, economic valuation of services supported by environmental flows is a promising way of bringing environmental flows to the decision-making agenda on equal terms with other water uses.

This Ph.D. project has developed a simple and transparent decision support tool for assessing various environmental flows scenarios and arriving at a negotiated environmental flows allocation and thereby a negotiated river condition and economic trade-off between water uses. The tool is based on an existing river basin simulation model, MIKE BASIN, and calculation procedures developed in MS Excel. The core of operationalising the tool is the development of the Service Provision Index (SPI). This approach explicitly links environmental flows to (socio)-economic values by deliberately focusing on ecosystem services. As such, it places due emphasis on the 'end product' of ecosystem functions to humans and renders environmental flows somewhat easier to justify and value. Furthermore, the SPI approach may be tailored to conform to case specific data availability. Therefore, it may be used as a desk-top method or a comprehensive holistic methodology, depending on the data and information available.

The SPI approach is potentially participatory and allows for stakeholders to be involved in several phases. While this is in line with the current trends of stakeholder-oriented water resources management, it requires commitment and resources beyond the scope of most environmental flows assessments. The case study presented in this thesis is an example of a rapid application of the SPI approach, where issues related to participation per se have not been addressed.

The main shortcoming of the SPI approach is that in data-scarce applications the links between flows and services are assumed. As more and more information becomes available, such links can be refined and documented. However, a major challenge remains regarding the establishment of links between flows and services: identifying the extent to which each flow class is responsible for service provision. Furthermore, water quality aspects of environmental flows should be incorporated into the SPI approach by linking SPI to water quality as well as to flow. Also concerning the links between services and values, several challenges remain. The assumptions of marginality and substitutability in the case of subsistence and large uncertainties are examples of challenges. While these are no excuse for not undertaking valuation, they may encourage the use of the total SPI instead of, or in addition to, total value when evaluating scenarios and providing decision support for IWRM. For example, the case study presented in the thesis highlights the shortcomings of using conventional economic valuation methods for estimating the value of services that supports the livelihoods of poor people. If poverty reduction is on the agenda in a river basin, SPI could serve as a better means of comparison between environmental flows scenario than economic value.

While many uncertainties and shortcomings remain, using SPI and economic valuation of environmental flows seems to be a promising way forward within the field of environmental flows assessment methodologies. In conclusion, this Ph.D. project has addressed the inherently multi-disciplinary link between environmental flows, ecosystem services and economic value and developed an operational tool for quantifying environmental flows in the context of Integrated Water Resources Management (IWRM).

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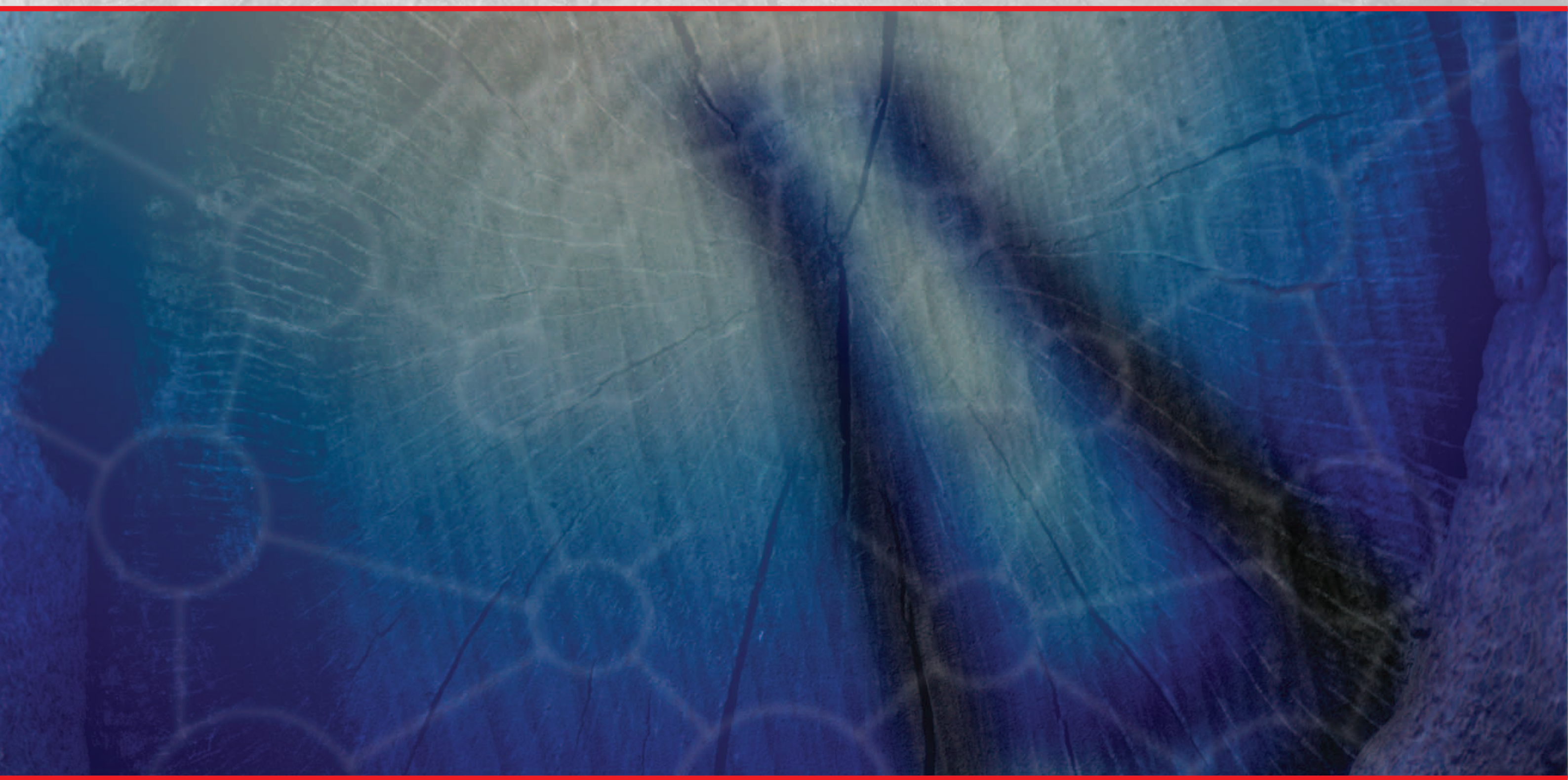

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