Wetland Health and Importance Research Programme

8

Wetland Valuation Volume I
Wetland Ecosystem Services and Their Valuation: A Review of Current Understanding and Practice

Authors: J Turpie, K Lannas, N Scovronick & A Louw
Series Editor: H Malan

WATER
RESEARCH
COMMISSION

TT 440/09

8 WETLAND VALUATION. VOL I

WETLAND ECOSYSTEM SERVICES AND THEIR VALUATION: A REVIEW OF CURRENT UNDERSTANDING AND PRACTICE

Report to the

Water Research Commission

by

Authors: J Turpie[1], K Lannas[2], N Scovronick[1] and A Louw[1]
Series Editor: H Malan[2]

[1] Anchor Environmental Consultants
in association with the
Percy FitzPatrick Institute, University of Cape Town

[2] Freshwater Research Unit,
University of Cape Town

PREFACE

This report is one of the outputs of the Wetland Health and Importance (WHI) research programme which was funded by the Water Research Commission. The WHI represents Phase II of the National Wetlands Research Programme and was formerly known as "Wetland Health and *Integrity*". Phase I, under the leadership of Professor Ellery, resulted in the "WET-Management" series of publications. Phase II, the WHI programme, was broadly aimed at assessing wetland environmental condition and socio-economic importance.

The full list of reports from this research programme is given below. All the reports, except one, are published as WRC reports with H. Malan as series editor. The findings of the study on the effect of wetland environmental condition, rehabilitation and creation on disease vectors were published as a review article in the journal Water SA (see under "miscellaneous").

An Excel database was created to house the biological sampling data from the Western Cape and is recorded on a CD provided at the back of Day and Malan (2010). The data were collected from mainly pans and seep wetlands over the period of 2007 to the end of 2008. Descriptions of each of the wetland sites are provided, as well as water quality data, plant and invertebrate species lists where collected.

An overview of the series
Tools and metrics for assessment of wetland environmental condition and socio-economic importance: handbook to the WHI research programme by E. Day and H. Malan. 2010. (This includes "*A critique of currently-available SA wetland assessment tools and recommendations for their future development*" by H. Malan as an appendix to the document).

Assessing wetland environmental condition using biota
Aquatic invertebrates as indicators of human impacts in South African wetlands by M. Bird. 2010.

The assessment of temporary wetlands during dry conditions by J. Day, E. Day, V. Ross-Gillespie and A. Ketley. 2010.

Development of a tool for assessment of the environmental condition of wetlands using macrophytes by F. Corry. 2010.

Broad-scale assessment of impacts and ecosystem services

A method for assessing cumulative impacts on wetland functions at the catchment or landscape scale by W. Ellery, S. Grenfell, M. Grenfell, C. Jaganath, H. Malan and D. Kotze. 2010.

Socio-economic and sustainability studies

Wetland valuation. Vol I: Wetland ecosystem services and their valuation: a review of current understanding and practice by Turpie, K. Lannas, N. Scovronick and A. Louw. 2010.

Wetland valuation. Vol II: Wetland valuation case studies by J. Turpie (Editor). 2010.

Wetland valuation. Vol III: A tool for the assessment of the livelihood value of wetlands by J. Turpie. 2010.

Wetland valuation. Vol IV: A protocol for the quantification and valuation of wetland ecosystem services by J. Turpie and M. Kleynhans. 2010.

WET-SustainableUse: A system for assessing the sustainability of wetland use by D. Kotze. 2010.

Assessment of the environmental condition, ecosystem service provision and sustainability of use of two wetlands in the Kamiesberg uplands by D. Kotze, H. Malan, W. Ellery, I. Samuels and L. Saul. 2010.

Miscellaneous

Wetlands and invertebrate disease hosts: are we asking for trouble? By H. Malan, C. Appleton, J. Day and J. Dini (Published in Water SA 35: (5) 2009 pp 753-768).

EXECUTIVE SUMMARY

Introduction

This study forms part of the scoping phase of the Water Research Commission-funded Wetland Health and Importance Research Programme (Phase II of the National Wetland Research Programme). The aim of this study was to review the wetland valuation literature, to ascertain how wetland valuation has been approached internationally, and how international and local experience can guide best practice for approaching wetland valuation in South Africa.

Why wetlands are valued

Wetlands are recognised as being valuable ecosystems which provide water, food and raw materials, services such as flood attenuation and water purification, and intangible values such as cultural and religious value. In some areas, they can be particularly important for peoples' livelihoods. Despite this, and legislation to protect them, they are increasingly threatened, with more than half the world's wetlands having been lost already. Wetlands are degraded beyond the socially optimal extent due to market failure (where markets do not reflect true values or costs) and government failure (perverse incentives, lack of well-defined property rights leading to open access and ignorance of decision makers as to the value of wetlands). Economic valuation helps to compare the real costs and benefits of ecosystem use and degradation, and allows more balanced decision-making regarding the protection and restoration versus degradation of wetlands. This facilitates optimal decision-making which maximises societal well-being.

Wetland services

Ecosystem services have traditionally been disaggregated into goods (= products), services (= ecosystem functions) and attributes (= structure, diversity, rarity, etc.). Under the recent Millennium Assessment they were classified into provisioning, regulating, cultural and supporting services. Provisioning services refers to the provision of goods such as water, food and raw materials. Regulating services are processes that contribute to economic production or save costs, such as flow regulation (including flood attenuation, regulation of base flows, groundwater recharge), sediment retention, water purification and carbon sequestration. Cultural services relate to ecosystem attributes and include the spiritual,

educational, cultural, recreational, existence and bequest value that is derived from use or appreciation of biodiversity. Supporting services are the biophysical process that underlie the first three, and should not be valued to avoid double-counting. There may also be disservices associated with wetlands, e.g. provision of breeding grounds for pests and pathogens. This section provides a review of the understanding of what factors influence the delivery of wetland services, as well as quantitative estimates of these services in biophysical terms.

Total Economic Value framework

Economic value can be defined in terms of peoples' willingness to pay for a commodity or state of the world. Net economic value can be expressed as the sum of consumer surplus and producer surplus. Value generated by ecosystems can be disaggregated into different types using the Total Economic Valuation framework: consumptive or non-consumptive direct use value, indirect use value, option value and non-use value.

Valuation methods

The methods used to value wetlands are no different from the methods used to value any other type of environmental asset. These include market value approaches (which rely on quantification of production), surrogate market or revealed preference approaches (which rely on observation of related behaviour) and simulated market or stated preference approaches (which rely on direct questioning). The simpler methods produce a total value, whereas those that involve construction of models are better for estimating marginal values (the additional value generated by each unit of production).

Consumptive and non-consumptive direct use values are generally estimated using *Market valuation*, based on estimates of quantities produced, prices and costs of inputs. Quantification of use can be complex if monitoring data are not available, and may involve key informant interviews, focus group discussions and household surveys involving detailed questionnaires about resource use. The *Change in production* approach involves constructing a model to estimate changes in the net benefits of the production of goods or services due to changes in the quantity or quality of inputs provided by the natural environment (i.e. marginal values).

Values associated with regulating services are typically measured using *Replacement cost* methods, for example the cost of building dams to replace a wetland's flood amelioration function. Alternatively, one can estimate the *damage costs avoided* due to the presence of the wetland, such as the damage that would occur due to flooding, or the *defensive expenditure* needed to prevent that damage, such as building dykes downstream. Marginal values can be estimated but requires an understanding of the relationships between wetland characteristics and their functioning.

Recreational value is measured in terms of tourism value and property value. Tourism value is measured using the *Travel Cost Method*. Data on money and time spent by users visiting a recreational site is used to construct a travel cost model from which a demand curve is derived. This enables estimation of value including consumer surplus. The accuracy of this method is hampered by complications such as multiple destination trips. The *Hedonic Pricing Method* uses linear modelling to isolate the contribution that the proximity or quality of an environmental asset makes to property prices. These models allow estimates of marginal value.

Non-use values can only be estimated using stated-preference approaches such as the *Contingent Valuation Method* (CVM). This involves a questionnaire survey in which a hypothetical question (or set of questions) is posed to respondents which elicits their willingness to pay for the preservation of biodiversity or their willingness to accept compensation for the loss of biodiversity. The method is controversial because it is prone to a number of biases. However, there are internationally accepted guidelines to minimise these biases. CVM is limiting because it usually only allows estimation of the impact of one or two changes. *Conjoint Valuation* allows the estimation of marginal value, by analysing responses to multiple scenarios. It involves generation of a model explaining how different attributes of an ecosystem contribute to its overall value, and the way in which this overall value changes when certain attributes change.

Benefits Transfer is the name for using results from other studies to estimate the value of similar areas under consideration. Although there are rules in doing this, most analysts have rejected the validity of this approach, due to the high degree of dissimilarity in both mean values and value functions between sites.

Putting values in perspective

There are various ways of expressing values to make them relevant to the decision-making context. They can be measured at a local to a national scale, and from a private (financial) or a social (economic) perspective.

Ecosystem values are generally expressed as a net economic value expressed as a current annual value. In many cases it is also very useful to consider the value of the wetland over a period into the future, especially where sustainability is an issue. This requires selecting a time frame and a discount rate which determines the weighting of future values. The values obtained over a series of years can be expressed as a single value, the 'net present value'. This value is obtained by discounting future values at a rate which is comparable with the interest obtained on alternative investments, and is the asset value of the wetland (equivalent to its purchase price). In predicting the future values, it is important to adjust for changes due to impacts on the ecosystem, otherwise an unsustainably used wetland will appear to be more valuable than it should.

It is sometimes useful to express values in a way that is compatible with national accounting systems. National Accounts quantify the value of production at a national scale and measure the total output in the economy (e.g. as Gross Domestic Product) and how this changes over time or under different policies. Many countries are now developing compatible natural resource accounts, for different resources and ecosystems. Two types of accounts are produced: production accounts, which measure value per year, and asset accounts which measure the net present value of the resource or ecosystem.

In some cases it is relevant to describe the contribution that wetlands make to poor households and people's livelihoods. Social Accounting Matrices can be used to assess the former. Estimating their contribution to people's livelihoods requires quantifying the other sources of household livelihoods as well. Nevertheless, simply estimating the proportion of household income generated by wetland resources does not necessarily provide an accurate idea of their importance in terms of risk spreading and safety-net function, and this needs to be described as far as possible (even qualitatively) through various appraisal methods.

Wetland valuation in practice

Wetland valuation is used to build local and political support for their conservation and sustainable use, to help diagnose the causes of environmental degradation and biodiversity loss, to allow more balanced planning and decision-making, and to develop incentive and financing mechanisms for achieving conservation goals.

Much of the early work was primarily to demonstrate that wetlands had high value. These included studies that were carried out to satisfy managers that the expense was worthwhile, to influence the development of wetlands policy, and to advocate the wise use of wetlands.

Valuation is increasingly applied in decision-making processes that evaluate the effects (costs and benefits) of alternative development options that affect wetlands. This often involves developing an understanding of the utility function underlying the wetland value, or comparing the value of alternative land uses (e.g. shrimp farming vs. intact mangroves). Valuation is now being applied in conservation planning studies to add a social dimension to what was previously a purely biodiversity issue.

Similarly, valuation of wetlands is fast becoming an integral component of water resource planning, and is used in the evaluation of alternative water allocation and environmental flow scenarios. Some of the most important early work on wetland valuation in the context of environmental flows was on the Hadejia-Jama'are floodplain in Nigeria, which demonstrated their importance in terms of agriculture, fishing and firewood as well as grazing lands for semi-nomadic herders.

There are some important contextual issues in valuation. In general, direct use values are normally considered at the local level, indirect use values at a broader scale, and non-use values at the broadest scale. Local-scale benefits may incur regional-scale costs, and vice versa. 'Local communities' have to be defined on the basis of explicitly stated criteria. In addition, the position of wetlands within a landscape has an influence on functioning and social setting, and therefore on the way in which valuation studies should be approached. Of particular importance is the property rights setting, as value will be influenced by the amount and type of access allowed. The heterogeneity of communities surrounding wetlands also has an important influence on how valuation studies are tackled, with greater heterogeneity requiring more sub-sampling. Finally, valuation studies conducted in a developing country context such as poor rural communal lands are approached somewhat

differently, having far more challenges to overcome, such as data, cultural and educational issues.

At the same time as methods have become increasingly refined, there has also been pressure to develop rapid, or cheaper, means of assessing the value of ecosystems. Benefits transfer, or use of existing estimates from other areas, has essentially been given the thumbs-down. To some extent, rapid rural appraisal methods offer a way of obtaining rough estimates of direct use value, for example by getting a group of people to demonstrate the relative value of different activities using piles of stones. Some authors have demonstrated that expert opinion of estate agents is just as good as the more data intensive hedonic pricing method for estimating the property value ascribed to ecosystems. In terms of indirect use values, the most expedient estimates appear to be those of replacement costs, which are less reliant on the physical quantification of ecosystem processes.

While different levels of confidence are acceptable for different types of decisions, it is particularly important that the confidence of the estimates is known to the decision-makers.

The use of non-monetary rapid assessment indices also provides an option for the rapid evaluation of the relative value of wetlands. WET-EcoServices is a recently-developed South African tool which scores wetlands in term of their capacity to provide ecosystem services, as well as the opportunity to provide the service. This is similar to an economic rationale of supply and demand. However the overall index is limited in its usefulness because of weighting and scale issues.

What are wetlands worth?

Much of the international work on the value of coastal wetlands and estuaries has concentrated on the value of mangroves. Studies in Africa have shown that floodplain wetlands are used extensively for harvesting fish, reeds, sedges, palm leaves, thatching grass, medicinal and food plants, and mangroves and salt at the coast. Some of these resources are used to manufacture a range of products such as sleeping bags, mats, baskets, bed ropes, hats, food covers, fans, ornaments, brooms and grain silos. The overall direct use values of these wetlands are correlated with the density of their surrounding inhabitants.

Numerous hedonic pricing studies, particularly in the developed world, have demonstrated a positive impact of wetlands on property value, though under certain circumstances, the reverse can be true. Travel cost studies have also shown the tourism value of wetlands to be substantial, including in South Africa. Several African wetlands have been shown to have negligible tourism value, whereas for others, such as the Okavango Delta, tourism value surpasses any other value.

There is extensive literature on the indirect value of ecosystem services. However, most studies highlight the difficulties in measurement of at least some components of indirect use value because of the considerable amount of biophysical information that is required. Much less work has been carried out on the non-use value of wetlands than on other types of value. The vast majority of studies have not arrived at a total economic value, and can only be considered partial valuations.

In southern Africa, wetland valuation studies have found a range of values from $47 200 to $80 900/ha for property value, $159-40 440/ha/y for tourism value, $1.4 to $378/ha/y for the value of harvesting resources and $28.35 to $5423/ha/y for ecosystem services. These ranges echo trends in the international literature. Indeed, meta-analyses of wetland studies have concluded that there are no predictive trends in value, which further strengthens the case that it can be difficult to estimate the value of a wetland based on studies of other wetlands.

Applying wetland valuation in South Africa

This review highlights some important lessons in applying wetland valuation in South Africa. South Africa has a multitude of wetland types, social contexts and they lie in a variety of geographic and landscape contexts. The problems facing South African wetlands are a mixture of those found in the developed and the developing nations. The decision making contexts, particularly regarding land use, conservation and development planning, and water allocation, are common problems in most of the countries where valuation has taken place. Being a developing country, data availability is often a constraint, and the lack of biophysical data on wetland functioning is probably one of the biggest obstacles to wetland valuation in South Africa.

There is no specific valuation context or situation that is peculiar to South Africa that has not been encountered in wetland valuation studies elsewhere. Thus, in general, wetland

valuation should continue to follow best practice for ecosystem valuation in general. Ideally this should continue until valuation studies can provide numerous examples of different types and sizes of wetlands in different geographic and social settings. Extrapolation of high-confidence values would require considerably more comprehensive valuation studies than exist at present.

Nevertheless, there is increasing pressure to develop rapid, cheaper methods in South Africa, particularly with the current emphasis on the determination of environmental flows under the South African National Water Act No. 36 of 1998, but also due to the pressures of development. Up till now, international experience has shown that the use of rapid methods is potentially fraught with inaccuracy, especially regarding the use of benefits transfer. However, there have been some promising studies which suggest that other rapid valuation techniques may be feasible, though these still require some level of data collection or surveys. If a desktop-level rapid valuation method is to be developed for South African wetlands at this stage, it will only be possible at a level that generates low confidence estimates, providing rough ranges of value suitable for coarse-level decision-making.

TABLE OF CONTENTS

LIST OF FIGURES

LIST OF TABLES

ACKNOWLEDGEMENTS

ABBREVIATIONS

BFI - Base Flow Index

C - Cost

CBA - Cost-benefit analysis

CBNRM - Community-based Natural Resource Management

CVM - Contingent Valuation Method

DEAT - Department of Environmental Affairs and Tourism

DWAF - Department of Water Affairs and Forestry

EC - European Commission

GDP - Gross Domestic Product

IMF - International Monetary Fund

IPCC - Intergovernmental Panel on Climate Change

NOAA - National Oceanic and Atmospheric Administration

NPV - Net present value

NRA - Natural resource accounts

OECD - Organisation for Economic Co-operation and Development

P - Price

Q - Quantity

SAM - Social accounting matrix

TCM - Travel-cost Method

TEV - Total economic value

UN - United Nations

WB - World Bank

WHI - Wetland Health and Importance (Research Programme)

WRC - Water Research Commission

1. INTRODUCTION

1.1 Background

1.1.1 The Wetland Health and Importance Research Programme

This study forms part of the Wetland Health and Importance (WHI) Research Programme, which falls under the National Wetlands Research Programme of the Water Research Commission (WRC). The objectives of the National Wetlands Research Programme are:

1. To initiate, support and manage research projects that contribute to wetland management.

2. To ensure the effective transfer of information on wetlands to institutions and persons involved in wetland management.

3. To promote human resource capacity in wetland management.

4. To ensure financial long-term sustainability of wetland research in South Africa.

This project forms part of the second of the three thrusts of the National Wetlands Research Programme:

- Phase I: Rehabilitation;
- Phase II: Wetland Health and Importance; and
- Phase III: Wise Use.

The Wetland Health and Importance (WHI) Research Programme is concerned with the development of rapid methods to assess the health and integrity of wetlands as well as their social importance and economic value. All of these aspects are vital for the effective management and protection of wetlands. Although there are techniques for the assessment of aquatic ecosystem health and socio-economic values have been developed or applied in South Africa, there are currently no definitive, well-developed methods (comprehensive or rapid) specifically designed for assessing *wetland* environmental condition, social importance and economic value.

The main aims of the WHI Research Programme are to:

1. Develop tools for assessing wetland ecological condition that will address the major needs of users in South Africa.

2. Develop tools for assessing wetland socio-economic importance that will begin to satisfy the needs of users in South Africa.

3. Develop a protocol to assess the loss of wetland function through degradation.

4. Implement a communication programme (including compilation of training modules) to

advise on the use of assessment techniques developed in the programme.

This study forms the resource economics component of the WHI research programme. Understanding socio-economic values of wetlands is important for management, conservation and development planning, and helps to justify investment in conservation or rehabilitation of wetlands. It will be an essential element of the determination of freshwater allocation to wetlands.

1.1.2 Rationale and aims of the Resource Economics component

Wetlands in South Africa are considered valuable from a biodiversity and ecosystem services perspective, but they are subject to numerous pressures including conversion, overexploitation, pollution and changes in hydrology. One of the reasons for this is that the economic consequences of management decisions are very poorly understood. Economic valuation of wetlands is increasingly being recognised as being a valuable aid to policy and decision making. Yet whereas valuation studies have been carried all over the world over the past two decades or so, very little work has taken place in South Africa. The international experience in wetland valuation can provide some useful lessons for the development of best practice in South Africa. In addition, those experiences, in conjunction with research conducted in southern Africa, will hopefully provide some of the insights required for the development of a relatively rapid method for the assessment of wetland values.

There is no standardised methodology for valuation of wetlands, and no guidelines for use in South Africa, although general guidelines for valuation have been developed under the Department of Water Affairs (DWAF) Classification Project (which does not considering wetlands *per se*), and guidelines are being developed under the WRC project on the valuation of goods and services of aquatic ecosystems for use in Resource Directed Measures (for determination of the freshwater reserve).

Current valuation methods are designed for comprehensive application, which means they are expensive. There is a need for more rapid methods to be investigated in terms of their feasibility for use, by assessing their relative accuracy and sufficiency for decision-making. However, in order to test the efficiency of a rapid method, it has to be compared with the results of a comprehensive assessment.

The overall objectives of the resource economics component are as follows:

1. Conduct a scoping study of methods to value wetland "goods and services";
2. Evaluate WET-EcoServices as a basis for determining the economic value of wetlands;
3. Develop a metric to assess socio-economic dependency; and
4. Develop a rapid wetland valuation protocol which takes into consideration the different types and geographical location of wetlands.

1.2 Aims of this study

This aims of this study were:

1. To provide an overview of the types of wetlands found in South Africa;
2. To review the different types of services provided by wetlands;
3. To review the quantification of the biophysical data required to estimate the value of wetland ecosystem services, and assess the availability of data in South Africa; and
4. To ascertain how the valuation of these services has been approached internationally, and how international and local experience can guide best practice for approaching wetland valuation in South Africa.

The findings of this study will form the basis for design of approaches for wetland valuation in South Africa for different decision-making contexts, including situations which call for rapid appraisal. The approach will be tested over the next two years, and will culminate in the development of a wetland valuation protocol.

1.3 Structure of the report

Chapter 2 provides the rationale for wetland valuation. Chapter 3 describes the types and distribution of wetlands in South Africa. Chapter 4 provides a detailed explanation of the ecosystem services provided by wetlands. Chapter 5 describes the valuation frameworks that typically guide wetland valuation at present, defining the concepts of ecosystem services and Total Economic Value. Chapter 6 reviews the different methods used in ecosystem valuation generally, all of which are also applied to wetland ecosystems. Chapter 7 explains the ways in which those values are expressed in order to put them in relevant perspective. Chapter 8 reviews valuation studies carried out for different purposes and how wetland valuation approaches and outcomes are influenced by geographic, social and decision-making contexts, and evaluates the usefulness of rapid assessment methods. Chapter 9 provides a review of wetland studies that have been carried out around the world,

summarising some of the results obtained, and focussing on the work done in southern Africa. A more detailed summary of results from international and African studies is provided in Appendix 1. Finally, Chapter 10 provides some brief remarks on how wetland valuation should be approached in South Africa. The report includes a comprehensive bibliography of literature relevant to wetland valuation.

2. WHY WETLANDS SHOULD BE VALUED

2.1 The importance of wetlands

Wetlands provide numerous goods and services to society, supporting millions of people around the world (Barbier *et al., 1*997). Indeed, the global value of wetlands and their associated ecosystem services has been estimated at US$14 trillion annually (Costanza *et al., 1*997). Wetlands provide rich wetland soils for agriculture, fish for sustenance, trees for timber and firewood, reeds for mats and thatching, as well as recreational opportunities. Rural households often harvest natural products for food, medicines, cosmetics or materials for shelter (Adaya *et al., 1*997, Barbier *et al., 1*997). In addition, the water itself is a valuable commodity. They provide services such as flood attenuation and water purification which benefit people far beyond the wetlands themselves. Wetlands also have less tangible values which may be linked to cultural heritage or religious values associated with them (Turpie *et al., 2*006a).

2.2 Factors leading to the degradation of wetlands

Wetlands are highly sensitive ecosystems which make them vulnerable to degradation (Turner *et al., 2*000). Despite their importance, and various forms of international and national legislation ratifying their protection (Bergstrom and Stoll, 1993), wetlands are highly endangered ecosystems which are increasingly becoming threatened (Barbier *et al., 1*997, Turner *et al., 2*000). It is estimated that since 1900 more than half of the world's wetlands have been destroyed and lost to other land uses (Barbier, 1993). Indeed, wetlands are frequently lost to development and other land uses which offer limited benefits or even end up being costly to the surrounding communities (Bowers, 1983; Turner *et al., 2*000). In South Africa, wetlands are lost in conversion to alternative land uses, or degraded due to overexploitation, pollution, invasion by alien plants and changes in hydrology (e.g. upstream water abstraction).

Several factors contribute to this trend including market failure associated with public good qualities of wetlands and externalities, and government failure associated with property rights, perverse incentives and distorted decision-making (de Groot *et al., 2*006; Vorhies, 1999; Stuip *et al., 2*002).

2.2.1   Public good qualities (market failure)

Many of the goods and services and amenity values provided by wetlands have the qualities of a public good; i.e. they seen as "free" and are thus not accounted for in the market (e.g. water purification or flood attenuation). When services are seen as free they tend to be wasted, or not accounted for in decisions which affect wetlands.

2.2.2   Externalities (market failure)

Another type of market failure occurs when markets do not reflect the full social costs or benefits of a change in the availability of a good or service (so-called externalities). Usually, those stakeholders who benefit from degrading an ecosystem are not the same as the stakeholders who bear the cost. For example, the price of agricultural products obtained from drained wetlands does not fully reflect the costs, in terms of pollution and lost wetland services, which are imposed upon society by the production process. The resulting loss of value (e.g. health, income) is not accounted for and the downstream stakeholders are generally not compensated for the damages they suffer (Stuip *et al., 2*002).

Because the functions that a wetland performs are often beneficial to people who do not necessarily live in the immediate vicinity of the wetland, their values are not always appreciated by property owners and do not provide a strong enough incentive to maintain wetlands rather than develop the land for other uses (Turner *et al., 2*000). Landowners usually do not consider the social values of wetlands such as waterfowl habitats, floodwater retention, groundwater recharge and nutrient filtration in the decisions they make in terms of land use (Danielson and Leitch, 1986). As a result, wetlands are often drained without regard for the optimum number of wetlands that should exist in terms of their value for society.

2.2.3   Perverse incentives (government failure)

Many policies and government decisions provide incentives (e.g. in the form of taxes or subsidies) for economic activity that often unintentionally work against the wise use of

wetlands, leading to resource degradation and destruction rather than sustainable management (Vorhies, 1999). For example, subsidies for shrimp farmers lead to mangrove destruction.

2.2.4 Lack of clear property rights (government failure)

One of the major problems in trying to conserve and protect wetlands is the fact that they are often open-access resources with limited control over how they are used and what is harvested from them (Turner *et al., 2*000; Helm, 1996). Ownership of wetlands can be difficult to establish. Wetland ecosystems often do not have clear natural boundaries and, even when natural boundaries can be defined, they may not correspond with an administrative boundary. Therefore, the bounds of responsibility of a government organization cannot be easily allocated and user values are not immediately apparent to decision-makers.

2.2.5 Lack of information (government failure)

Many sectors of society view wetlands as being of little or even of negative value (Woodward and Wui, 2001; Turner *et al., 2*000). Incomplete knowledge of the economic and ecological importance of wetlands leads to unsustainable land practices or development taking place (Adaya *et al., 1*997). The economic benefits and services provided by wetland ecosystems are frequently overlooked by governments, developers, private industry and other land users (Emerton, 1998). Lack of information can thus result in distorted decision-making.

2.3 What valuation can achieve

Economic valuation helps to give an indication of the real costs and benefits for ecosystem use and degradation (Pearce *et al., 1*994), and allows more balanced decision-making. It provides a basis for quantifying the benefits that people receive from wetlands, the cost incurred from their loss and the relative profitability of sustainable land practices and resource harvesting from wetlands compared to other more destructive activities (Emerton, 1998). Environmental economic valuation endeavours to place a monetary value on the goods and services provided by an ecosystem in an attempt to compare the benefits of preservation with those obtained through development (Batie and Shabman, 1982). It facilitates optimal decision-making which maximises societal well-being (Batie and Shabman, 1982), as well as promoting policies which protect the environment (Helm, 1996).

As a result of an increased understanding of the services provided by wetlands, many that have been converted to other land uses in the past are now being restored at high cost (Stuip *et al.,* 2002). A prior understanding of the impacts of these developments would have been far more efficient.

3.   WETLANDS IN SOUTH AFRICA: DISTRIBUTION AND TYPES

3.1   Introduction

This section provides a brief background on the wetland regions and types of wetlands found in South Africa, the classification of wetlands, and their distribution in South Africa.

3.2   Wetland eco-regions in South Africa

Cowan (1995) defined twenty-six wetland regions in South Africa (including Lesotho and Swaziland) based on topography, hydrology and nutrient regimes (Figure 3.1).

These regions can be grouped into four broad groups based on geomorphology and climate:
1. Plateau;
2. Mountains;
3. Coastal slopes and rimland; and
4. Coastal plain.

The subdivisions within these groups are determined by differences in geology.   Each wetland region has characteristic types of wetlands (Table 3.1).

3.3   Types of wetlands considered and adoption of a wetland typology

Wetlands are defined in the Ramsar Convention as

> "*areas of marsh, fen, peatland or water, whether natural or artificial, permanent or temporary, with water that is static or flowing, fresh, brackish or salt, including areas of marine water the depth of which at low tide does not exceed six metres*" (Ramsar, 1971).

In the National Water Act No 36 of 1998, wetlands are defined as

> "*Land which is transitional between terrestrial and aquatic systems where the water table is usually at or near the surface, or the land is periodically covered with shallow*

water, and which land in normal circumstances supports or would support vegetation typically adapted to life in saturated soil".

This study is concerned with wetlands in the narrower sense in that it does not include lakes, rivers, estuaries or marine systems, and does not include artificial wetlands.

The characteristics of a natural wetland are determined by the interaction of the quality and quantity of inflows and outflows, the geology, soils and topography, the climate and how they are used or managed (Palmer *et al., 2*002). Wetlands may receive water from rainwater, surface water, groundwater or a combination of these. They lose water through evaporation, evapotranspiration, surface flows or groundwater flows. These variations lead to a variety of types of wetlands that differ in their characteristics and functioning.
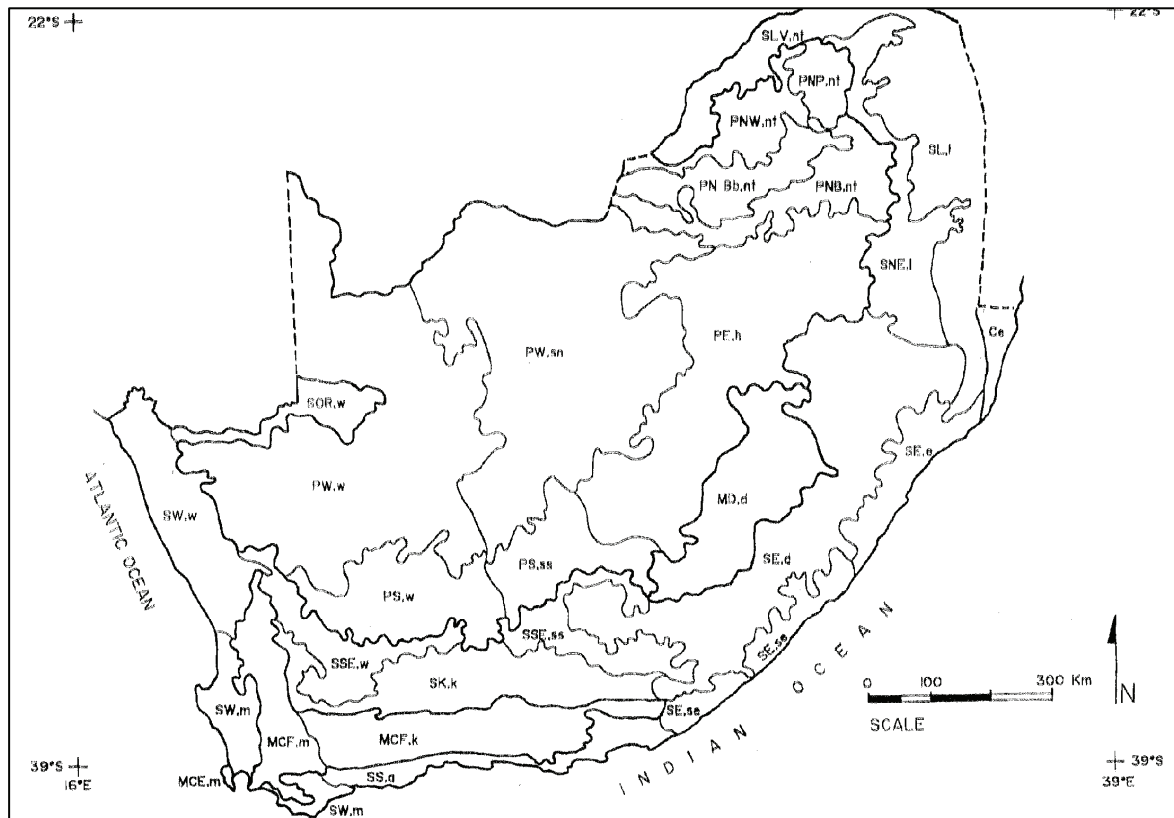
Figure 3.1: Wetland regions of South Africa (Cowan, 1995)

Table 3.1: Wetland regions described by Cowan (1995), typical wetland types found in the region and some well-known wetlands

Region	Typical wetlands	Examples
1. Plateau		
1. Western plateau, desert region	Pans and deflation basins	
2. Western plateau, steppe region	Pans	
3. Southern plateau, desert region		Grootvloer, Verneukpan, Van Wyksvlei
4. Southern plateau, steppe region	Pans	
5. Eastern plateau, highveld region	Riparian grass and reed marshes, and numerous pans	Wakkerstroomvlei, Seekoeivlei at Memel, Blesbokspruit
6. Bankenveld, N Tvl region	Riparian reed swamps	
7. Waterberg, N Tvl region	Seeps and small reed marshes (vleis)	
8. Bushveld basin, N Tvl region		Nylsvlei
9. Pietersburg Plateau, N Tvl region	Riparian wetlands	
2. Mountain		
1. Drakensberg/Maluti highlands	Alpine bogs, fens, restio marshes, grass marshes	
2. Cape Fold Mtns, Karoo region		Gamtoos floodplain
3. Cape Fold Mtns, Mediterranean region	Restio marshes	
3. Coastal slopes and rimland		
1. W coastal slope, desert region	Coastal pans and salt marshes	Orange R mouth wetland, Olifants River floodplain
2. W coastal slope, Mediterranean region	Coastal lakes, salt marshes, hygriphilous fynbos and restio marshes	Verlorenvlei, Berg R estuary, Langebaan Lagoon, Bot R mouth, Heuningnes estuary, De Hoop
3. S escarpment, desert region		
4. Karoo, karoo region	Pans	
5. S escarpment, s steppe region	Grass vleis, seeps, sedge marshes	
6. S coast, temperate region	Coastal lakes	Wilderness lakes
7. E coastal slope, Drakensberg region	Grass and restio marshes, reed swamps	Blood River Vlei, Mvoti Vlei, Hlatikulu Vlei, Franklin Vlei
8. E coast, se coastal region	Saltmarshes common in estuaries, mangrove swamps	
9. E coast, subtropical region	Lagoons, reed marshes, swamp forest and mangrove swamps	
10. N escarpment, lowveld region	Diverse, pans, grassland vleis	Lake Chrissie, Steenkampsberg vleis
11. Lowveld, lowveld region	Rivers with distinctive riparian communities	Wambiya pans, Levuvhu floodplain
12. Limpopo valley, N Tvl region		Limpopo floodplain and related pans
13. Orange R canyon, desert region	Small riparian reed swamps	
4. Coastal Plain		
1. Coastal plain, subtropical region	Floodplains, swamp forest, swamps, hygriphilous grass wetlands, coastal lakes, coral reefs.	St Lucia, Lake Sibaya, Kosi System, Muzi swamps, Pongola floodplain

Shaded areas are largely estuarine

South African wetlands have been **classified** in various ways (e.g. Noble and Hemens, 1978; Breen and Begg, 1989; Cowan, 1995; Jones, 2002; Palmer *et al., 2*002; Kotze *et al., 2*008), and are mostly based on hydrogeomorpic characteristics. Some classification systems distinguish **palustrine wetlands, riverine wetlands** and **pans** (e.g. DEAT; Table 3.2). Several authors (e.g. Palmer *et al., 2*002) make a primary distinction between **seeps, floodplains** and **pans**. The Wetland Health and Importance Research Programme has adopted the classification system developed by Kotze *et al.* (2008), which distinguishes **seeps, valley-bottom wetlands, floodplains** and **pans**. For the types of wetlands of relevance to this study, this classification is similar to that of Ewart-Smith *et al.*'s (2006) more comprehensive classification system, except that it does not include "depressions linked to streams", which is a rarely occurring wetland type. Kotze *et al.* (2008) identified four main wetland types (or six types including subdivisions), based purely on hydrogeomorphic characteristics. The wetland types mentioned above are described in more detail below, and the characteristics of the types classified by Kotze *et al.* (2008) are summarised in Table 3.3**:** Wetland hydrogeomorphic (HGM) types typically supporting inland wetlands in South Africa..

Pallustrine wetlands are all non-tidal wetlands dominated by persistent emergent plants (e.g. reeds) emergent mosses or lichens, or shrubs or trees (see Cowardin *et al., 1*979).

Riverine or **riparian wetlands** are wetlands adjacent to a stream or river that are influenced by stream-induced or related processes.

Seeps or mires (also known as peat accumulating wetlands, or hillslope seepage wetlands) are most common in alpine areas or catchment source areas. These are further subdivided into bogs and fens (Schwabe, 1995). Bogs are isolated systems that have no major streams entering or leaving them and are generally found on the moister south-facing slopes, and are dominated by short sedges and grasses with hummocks. They receive water from rainfall and groundwater. Fens are much larger and also tend to have large lawns of sedges and grasses, but are found on the warmer north-facing slopes and are not as water-logged. They receive water from streams or groundwater, and tend to discharge their water into streams.

Table 3.2: Terminology and primary classification of non-tidal wetlands by different authors, and the approximate correspondence between these types
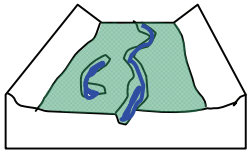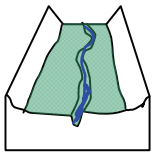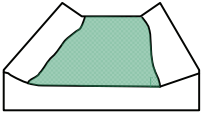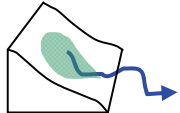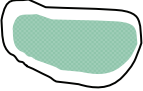
Brinson, 1993	DEAT (in litt, 2008)	Palmer *et al.,* 2002	Kotze *et al.,* 2008
Peatlands	Pallustrine wetlands	Seeps or mires (subdivided into bogs and fens)	Hillslope seepage wetlands (with and without linkage to a stream)
			Valley-bottom wetlands
Riverine wetlands	Riverine or riparian wetlands	Floodplains	Floodplain wetlands
Depressional wetlands	Pans	Pans	Pans

Valley bottom wetlands are palustrine wetlands that may or may not have a well defined stream channel but lack characteristic floodplain features. They receive water from overtopping of a main channel entering the wetland and from adjacent slopes. Depending on the slope of the wetland, they may be characterised by net removal or accumulation of sediments.

Floodplains are riparian wetlands that occur adjacent to river channels, tend to have a linear form (Rogers, 1995). They receive water from the river during higher flow events and lose water back to the river downstream. Energy and materials from surrounding landscapes converge and pass through riparian wetlands in greater amounts per unit area than in any other ecosystem (Rogers, 1995). Floodplains can contain riparian marshes and swamps that are permanently inundated, river-source sponges that have perennially saturated soils, and grasslands that are seasonally or intermittently inundated and saturated (Rogers, 1995). Floodplains that contain standing water bodies such as backwater swamps or oxbow lakes are known as storage floodplains. An inland delta such as the Okavango delta in Botswana would also be described as a floodplain, with the main difference that most water is lost through evaporation and via seepage into the groundwater rather than to downstream ecosystems.

Pans tend to be shallow depressions which receive water from precipitation and lose it via evaporation. They range from pans that are permanently or seasonally inundated in higher rainfall areas to pans that remain dry for years in the more arid regions. Permanently inundated pans can be likened to lakes (Allan *et al., 1*995). Pans are common throughout South Africa, especially in flatter areas of the grassland, Nama Karoo and Kalahari, and range from under a hectare to over 1000 ha in size (e.g. Barberspan; Allan *et al., 1*995).

Table 3.3: Wetland hydrogeomorphic (HGM) types typically supporting inland wetlands in South Africa. Contribution of the water source is described as: * usually small, ** usually large, or */ *** may be small or important depending on the local circumstances. Source: Kotze *et al.,* 2008.

Hydrogeomorphic types	Description	Source of water maintaining the wetland[1]	
		Surface	Sub-surface
Floodplain	Valley bottom areas with a well defined stream channel, gently sloped and characterized by floodplain features such as oxbow depressions and natural levees and the alluvial (by water) transport and deposition of sediment, usually leading to a net accumulation of sediment. Water inputs from main channel (when channel banks overspill) and from adjacent slopes.	***	*
Valley bottom with a channel	Valley bottom areas with a well defined stream channel but lacking characteristic floodplain features. May be gently sloped and characterized by the net accumulation of alluvial deposits or may have steeper slopes and be characterized by the net loss of sediment. Water inputs from main channel (when channel banks overspill) and from adjacent slopes.	***	*/ ***
Valley bottom without a channel	Valley bottom areas with no clearly defined stream channel, usually gently sloped and characterized by alluvial sediment deposition, generally leading to a net accumulation of sediment. Water inputs mainly from channel entering the wetland and also from adjacent slopes.	***	*/ ***
Hillslope seepage linked to a stream channel	Slopes on hillsides, which are characterized by the colluvial (transported by gravity) movement of materials. Water inputs are mainly from sub-surface flow and outflow is usually via a well defined stream channel connecting the area directly to a stream channel.	*	***
Isolated hillslope seepage	Slopes on hillsides, which are characterized by the colluvial (transported by gravity) movement of materials. Water inputs mainly from sub-surface flow and outflow either very limited or through diffuse sub-surface and/or surface flow but with no direct surface water connection to a stream channel.	*	***
Depression (includes Pans)	A basin shaped area with a closed elevation contour that allows for the accumulation of surface water (i.e. it is inward draining) and/or intersection of groundwater. It may also receive sub-surface water. An outlet is usually absent, and therefore this type is usually isolated from the stream channel network.	*/ ***	*/ ***

[1] Precipitation is an important water source and evapotranspiration an important output in all of the above settings

Pans vary in salinity, with many larger pans being saline. They can also range from being freshwater systems during the wet season to saline systems as evaporation proceeds during the dry season. The more saline pans tend to be less vegetated than the freshwater pans, which support reed and sedge marshes.

The terminology to describe wetland characteristics can be confused with the wetland typology, since the association varies. In general, flood- or other **plains** are typically grassy, **marshes** are dominated by emergent vegetation such as sedges and reeds, and **swamps** refer to any kind of wetland, typically ranging from floodplains and marshes to waterlogged forest rather than open pans. Note that a *vlei* is a colloquial South African term for a wetland, but more often than not describes a vegetated pan or floodplain that tends to contain some open water at least seasonally.

3.4 The distribution of wetlands in South Africa

DEAT provides basic information on the distribution of different types of wetlands in South Africa. There are no other comparable maps available which relate directly to the other classifications described above. While pans tend to be widespread and particularly common on the interior plateau, riverine and palustrine wetlands are strongly associated with the higher rainfall and mountainous areas of South Africa (Figure 2.2). Riverine wetlands tend to be associated with the lower slopes of the Drakensberg and Cape mountains and major river systems.

A variety of more detailed maps of wetlands have been produced at local and regional or provincial scales. The National Wetlands Inventory project is currently underway and will provide a relatively detailed mapping of wetlands at a national scale. Using the National Landcover 2000 map as a starting point, and collating existing data from other projects, the project has mapped over 114 000 wetlands (Figure 3.3). The beta version of the National Wetlands Map was released in 2006, and has since been updated twice (version II in 2008, version III in 2009).
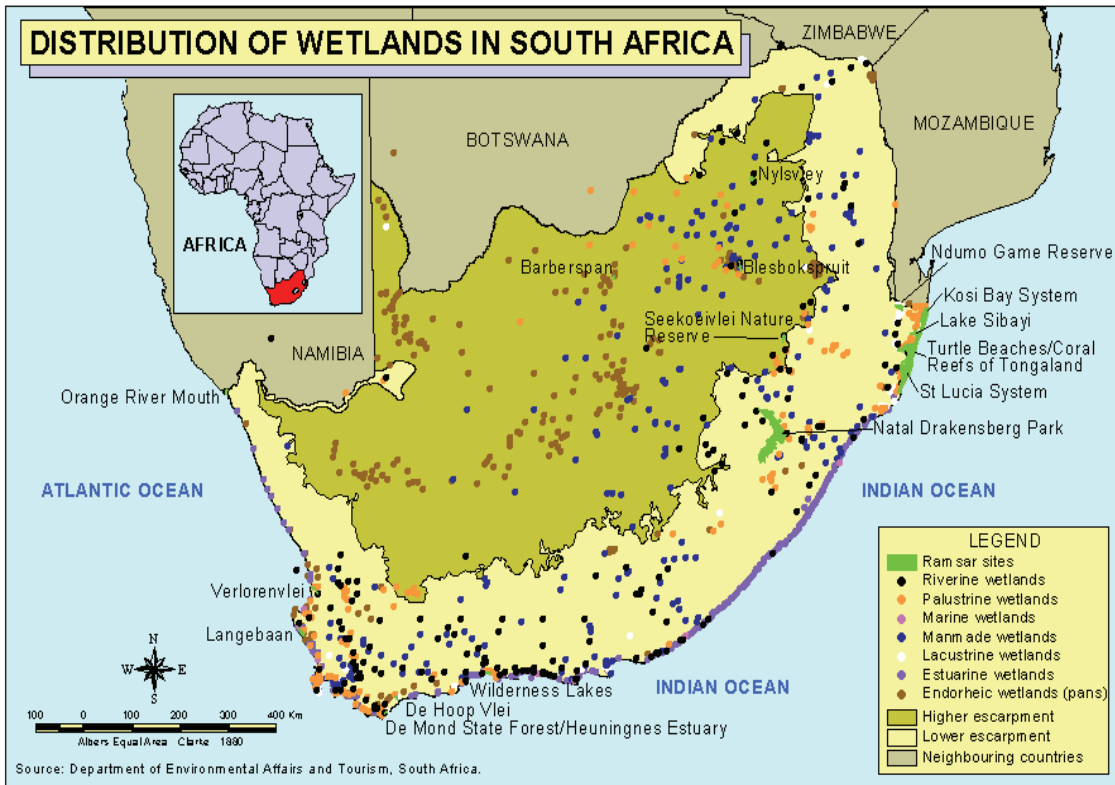
Figure 3.2: Distribution of different types of wetlands in South Africa (Source: DEAT).
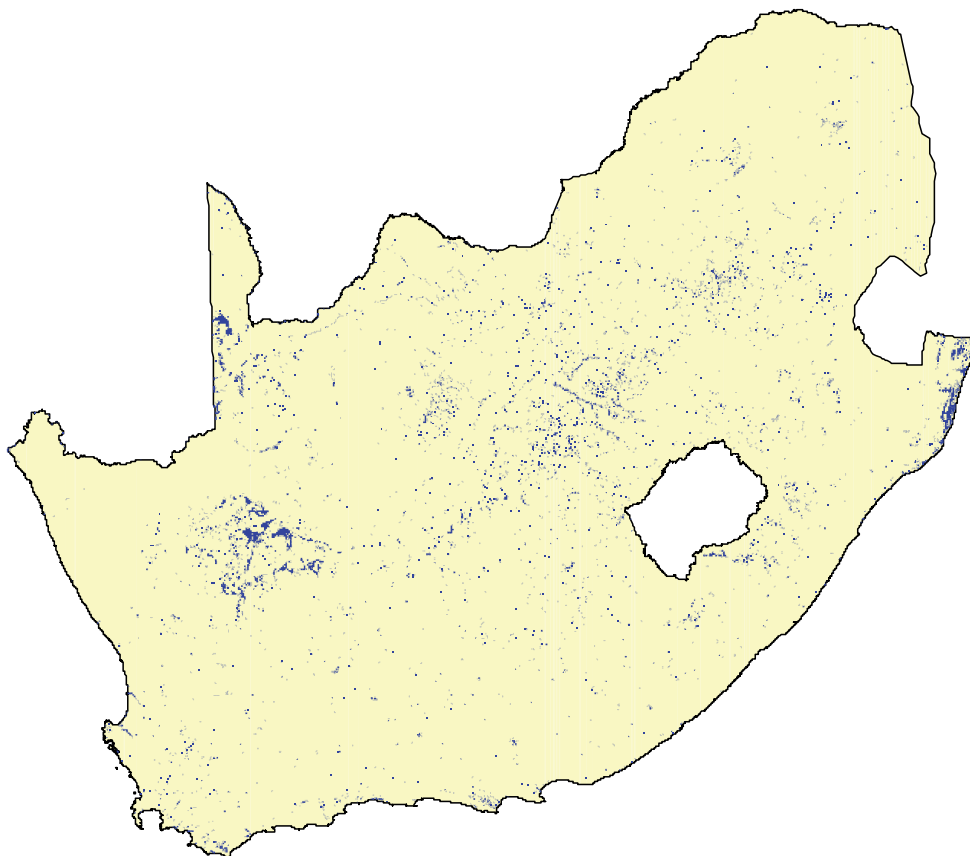
Figure 3.3: Wetlands of South Africa (source: SANBI), excluding identified large reservoirs.

4. WETLAND ECOSYSTEM SERVICES

4.1 Concepts of ecosystem services

Wetlands, like other ecosystems, offer a range of **goods, services and attributes** that generate value and contribute to human welfare (Barbier, 1994). The concept of ecosystem goods and services, popularised in the ecological-economics literature, stems from the perception of ecosystems as natural capital which contributes to economic production.

Goods, services and attributes may be defined as follows:
- **Goods** are harvested resources, such as fish;
- **Services** are processes that contribute to economic production or save costs, such as water purification; and
- **Attributes** relate to the structure and organisation of biodiversity, such as beauty, rarity or diversity, and generate less tangible values such as spiritual, educational, cultural and recreational value.

Goods, services and attributes are often referred to collectively as 'ecosystem services', or 'ecosystem goods and services'. However this often results in the value of ecosystem attributes being overlooked by those who are not aware of this.

More recently, the Millennium Ecosystem Assessment (2003) categorized the services obtained from ecosystems as follows:
- **Provisioning services** such as food and water;
- **Regulating services** such as flood and disease control;
- **Cultural services** such as spiritual, recreational, and cultural benefits; and
- **Supporting services**, such as nutrient cycling, that maintain the conditions for life on Earth.

The first three align well with the definitions of goods, services and attributes described above. The fourth, supporting services, has created some controversy as inclusion of these 'services' in a valuation study can lead to double counting. It does, nevertheless, highlight the fact that the other services cannot be generated without these underlying processes.

4.2   Ecosystem services associated with wetlands

The main types of ecosystem goods, services and attributes that would be associated with aquatic ecosystems are described in Table 4.1.

Table 4.1: Types of services provided by inland wetlands, based on Costanza *et al., 1*997 and the Millennium Assessment (2003)

Types of Services		Description
Provisioning services	Water	Provision of water for livestock or domestic use
	Food, medicines	Production of wild foods and medicines
	Grazing	Production of grazing for livestock
	Raw materials	Production of fuel, craftwork materials, construction materials
	Genetic resources	Medicine, products for materials science, genes for resistance to plant pathogens and crop pests, ornamental species
Regulating services	Climate regulation	Carbon sequestration. Wetlands are believed by some to be carbon sinks that contribute towards reducing carbon emissions
	Water regulation	Flood attenuation − Reduction of the amplitude and velocity of flood waters by wetlands, reducing downstream damage
		Groundwater recharge − Differential recharge to groundwater relative to surrounding vegetation types
		Dry season flows − Moderating the seasonality of downstream
	Sediment retention	Retention of soil and fertility within an ecosystem
	Waste treatment	Breaking down of waste, detoxifying pollution; dilution and transport of pollutants
	Regulation of pests and pathogens	Change in ecosystem health affects the abundance or prevalence of malaria, bilharzia, liver fluke, black fly, invasive plants, etc.
	Refugia	Critical breeding, feeding or watering habitat for populations that are utilised elsewhere.
Cultural services	Abundance, rarity and beauty of species, habitats and landscapes	Providing opportunities for : Cultural activities and heritage; Spiritual and religious activities and wellbeing; Social interaction; Recreational use and enjoyment; and Research and education.

Current understanding of the characteristics and processes that give rise to these services, and the way in which they are assessed or measured, is summarised below, with emphasis on understanding of South African wetland services.

4.3   Provisioning services

4.3.1   Natural resources

Wetlands provide a store of freshwater that can be used for domestic purposes or for watering livestock. Several kinds of living (e.g. reeds, thatching grass, firewood, fish) and non-living resources (e.g. clay) are harvested from wetlands for food, medicine and raw materials. Wetlands are also commonly used as grazing areas, especially during the dry season and are known to have a higher grazing potential than surrounding uplands.

The use of these resources can be quantified on the basis of data collected using the social survey methods described in a later section. Biophysical data are not necessarily required to estimate current value, but will be useful when conducting a rapid assessment. However, they are required in order to estimate net present value, which takes sustainability of use into account. For renewable resources, this would require an assessment of the current harvest relative to the rate of production of the resource. There are few, if any examples of doing this quantitatively in the literature. For some resources such as fish, stock and yield assessment is extremely difficult. A short-cut method has been provided by Welcomme (1985) based on an analysis of fish catches for floodplains of different sizes. Welcomme (*op. cit.*) found a relatively constant yield per unit area of floodplain wetlands, and this could be taken as a rough guide of the level of sustainability of the current harvest. There are also other short-cut methods for developing indices of levels of sustainability of inland fisheries in general, such as Rapfish (Pitcher & Preikshot, 2001), which uses a variety of social, ecological and economic indicators that can be assessed at a desk-top level in order to rate the level of sustainability of the fishery. For other resources, e.g. reeds, the absence of information on sustainable yields relative to stocks is more surprising. It should be fairly easy to estimate yields as a function of standing stocks or (preferably) area, based on understanding of the biology of the species.

Biophysical data would also be required in order to estimate the change in the productivity and availability of these resources as a result of changes in wetland characteristics or functioning that affect the degree to which they can be used. There are few examples of studies that attempt to do this, however, especially within southern Africa. Most of these are emerging in the environmental flows literature. The environmental flow assessment for the Lesotho Highlands Water Project was one of the pioneering attempts to estimate change in the value of resources harvested as a result of changes in water management. This was achieved by biologists estimating the percentage change in each resource (as a range of possibilities) under each of a number of scenarios. These estimates were then used as the

basis for estimating the change in value. An ongoing study on environmental flows of the Pangani River Basin in Tanzania has involved development of models that automate this process, allowing the analysis of any number of scenarios. What these studies have in common are that they are based on one or two site visits in which limited information is collected on the current state of the resource base, and the estimated changes are based on expert opinion. In the Pangani River Basin example, the assumptions made are formalized in explicit response curves. In many cases actually quantifying those response curves with statistically meaningful data might take many years.

4.3.2   Grazing

Grazing capacity is usually higher in wetlands than in surrounding upland areas, and can be more than double that of the upland areas (Turpie *et al., 1*999; Lannas and Turpie, 2009). This is relatively easily assessed by comparing the stocking rates in upland versus wetland areas.

4.3.3   Genetic resources

Ecosystems provide genetic resources which are sought after by bio-prospectors for medicinal purposes, for the development of horticultural varieties or for the improvement of crops. This value is related to the genetic diversity of an ecosystem type. Wetlands are not particularly valued in this sense, but would have some potential to provide genetic resources. Although it is reasonable to assume that wetlands with higher diversity would be more valuable in this regard, any attempt to put a number to this can only be pure guesswork.

4.4   Regulating services

4.4.1   Carbon sequestration

Climate change caused by increases in the emissions of greenhouse gases will carry a cost of about 2-7% of Gross Domestic Product (GDP) by 2050 (Fankhauser and Tol, 1997), due to changes in ecosystem productivity, ecotourism opportunities, disease vectors, agricultural production and due to infrastructural damage, among other effects (Turpie *et al., 2*004). The sequestration of carbon by ecosystems is thus considered an important service, which offsets the damage caused by increasing atmospheric carbon and resultant global climate change.

Carbon is sequestered when it is taken up by plants in the growth process and stored in above and below-ground plant biomass and peat deposits. In addition, litter production and other processes lead to the accumulation of carbon in soil. The amount stored in plant biomass is a relatively constant function of total mass, but the rate of carbon uptake from the atmosphere depends on the growth rate of these plants. The amount stored in soils differs according to vegetation cover and land use.

While it is relatively straightforward to determine the standing stock of carbon in a landscape, estimating the rate of carbon sequestration is a more complex issue. This is related to the rate of carbon uptake, and also to how permanently the carbon is stored. In terms of carbon trading, only the restoration of long-lived indigenous trees is considered valid. Nevertheless, faster growing vegetation may result in high levels of soil carbon sequestration, even if biomass carbon is not stored for long.

There is a rapidly-expanding literature on carbon sequestration. Tropical forests have received most attention as terrestrial carbon sinks due to their high CO_2 sequestration capacity (260-430 tons C/ha; Behling, 2002). Despite their relatively low rate of sequestration, savannas are considered important because of the large area they occupy (San Jose and Montes, 2001). In the case of savannas, carbon sequestration rates have been found to be highly variable seasonally and annually, depending on ecosystem health and fire regimes (Abril and Bucher, 2001; Kirschbaum, 2003). In Australia, savannas have been estimated to sequester carbon at a rate of 0.5 to 1.5tons/ha/y (Beringer *et al.,* 2007). In general the carbon sink capacity is linked to vegetation biomass. However, no comparable estimates are available for wetlands. Turpie *et al.* (2006b) had to base their estimate of the carbon sequestration value of the Okavango delta on the sequestration rates of terrestrial vegetation and the relative biomass of the different vegetation types of the delta. This led to estimates of 1 to 1.4tons/ha/y for different parts of the delta (Table 4.2). This did not take into account the degree to which the carbon may be recycled into the atmosphere by various processes. For example, it has been estimated that savannas sequester carbon at a rate of 192Tg C/y, but when emissions (e.g. from burning) are taken into account the net rate is about 17.5Tg C/y (San Jose and Montes, 2001).

Table 4.2: Estimated rates of carbon sequestration for the Okavango delta based on sequestration rates for Australian rangelands and differences in vegetative biomass (Turpie *et al.,* 2006b)

Wetland component	Carbon sequestration rate (T/ha/y)
Permanently-flooded area	1.08
Regularly-flooded area	1.35
Seasonally flooded	1.39
Occasionally flooded	1.36
Rarely flooded	1.00

Estimating carbon sequestration values for wetlands may be misleading in terms of estimating the value of wetlands for climate regulation. In a review of wetland dynamics and their contribution to greenhouse gas emissions, Cao *et al.* (1996) describe the complex interplay between carbon and methane (a more potent greenhouse gas than carbon). Despite the fact that wetlands store soil carbon substantially more than their comparable global land coverage (Bergkamp and Orlando, 1999), much of this benefit may be offset by the relatively high natural methane emissions produced from decomposition under anaerobic conditions characteristic of wet environments. In addition to natural factors that impact storage and emissions of carbon and methane, such as geography, climate and wetland type, land-use also impacts the two greenhouse gases, and often in opposite ways. One example would be the draining of a wetland for alternative land use. Under this scenario, methane emissions would likely decrease, but the accumulated soil carbon could be quickly released. The interplay of multiple factors influencing the source-sink carbon dynamics of wetlands may be one reason that wetlands have received relatively little attention compared to other ecosystems in terms of sequestration. In general, the role of wetlands in the global carbon cycle is not well understood (Bergkamp and Orlando, 1999).

4.4.2 Flow regulation

Flood attenuation occurs when wetlands ameliorate the potential impacts of flood events by absorbing the flood peaks and lengthening the flood period at a lower level, thereby reducing the risk of damage caused by flooding downstream. Flood attenuation occurs due to the detention storage and vegetative resistance to flow through an area (Ogawa and Male, 1986). A wetland that attenuates a flood effectively will have a broader flatter peak on the flood hydrograph. Flood attenuation results in reduced flow rates downstream, and hence a reduction in bank and streamed erosion as well as reduced risk of flooding of downstream areas. **Groundwater recharge** occurs when wetlands serve to intercept precipitation during the wet season, so that it infiltrates into the ground rather than flowing downstream. This

water may be utilised elsewhere via boreholes or wells (Thompson and Goes, 1997, in Acharya, 2000; Turpie *et al., 2*006b) or may be released lower down in the catchment over a delayed period, thereby helping to **augment base flows** (the portion of river flow that comes from subsurface sources) during the dry months (Turpie and van Zyl, 2002). Dry-season flows are critical to aquatic ecosystem health, as well as to rural populations that are directly reliant on rivers or springs for agriculture, domestic use and livestock watering. In addition, wetlands are also widely understood to act as holding areas, or 'sponges' that capture water during the wet season and release it over a delayed period, thereby potentially **enhancing dry season flows** (Adamus and Stockwell, 1983, Siegel, 1988, Thompson and Hollis, 1995), though this is contested (see below).

There has been relatively little quantitative description of the hydrological functions performed by wetlands (Smakhtin and Batchelor, 2005).

At its simplest, the flood attenuation function can be modelled by treating the wetland as a bucket that can be filled till it overflows. A similar approach was used by Krasnostein and Oldham (2004), but accounting for outflows as well – also known as a 'leaky bucket' model (Jothiyangkoon *et al., 2*001).

The type of soil and its porosity also plays a role in flow regulation through its effect on holding capacity (Carter *et al., 1*979). Ming *et al.* (2007) evaluated wetland flood attenuation capacity by how much water was required to saturate the wetland, taking the soil porosity into account. Core samples were taken at various sites across the study area and the porosity of the soil samples were measured as the total space between the samples when dried. The naturally present water content was measured as well and this figure was subtracted from the soil to calculate how much space would be available for additional water storage under inundated conditions. It is important to note, however, that a wetland that is drained, and therefore less saturated, may have more holding capacity, but a flood will move through it quicker, and this will not attenuate the peak to the degree of a partially saturated wetland (Woltemade and Potter, 1994).

Flood attenuation may be more a function of the resistance of the wetland, rather than its holding capacity (Woltemade and Potter, 1994). Resistance is related to vegetation cover, and is measured using Manning's co-efficient of roughness, originally developed in civil engineering to measure water velocity through artificial channels (Woltemade and Potter, 1994). Water storage and flood attenuation tend to be greater in wetlands with substantial

water level fluctuations, such as those with large wet meadow zones or with intermittent, seasonal, temporary, or semi-permanent hydrologic regimes (Ming *et al., 2007*).

In South Africa, Smakhtin and Batchelor (2005) used daily stream flow records (1971-1997) from above and below the Rustenberg Nature Reserve wetland (about 3 km by 150 m) to describe the flow regulation functions of the wetland. These functions could not be described by comparing the outflows with the inflows, since there were other flow inputs between the two gauges. Thus a no-wetland situation needed to be simulated for comparison. Smakhtin and Batchelor (2005) considered various options for approaching the problem. Using a rainfall-runoff model in which the area is divided into homogenous subareas with rainfall inputs, a model could be created and calibrated to present day conditions with wetland, and then run without the wetland (by modifying characteristics of the wetland subarea(s). This could not be achieved because of (a) a lack of rainfall data from appropriate locations and (b) daily data would not have sufficient resolution to simulate instantaneous maximum daily flows. Another option was to use flood-routing methods, which examine how a flood hydrograph is modified as storm water flows downstream due to resistance along the way. This method generally requires a time period of less than 1/6th of the time to peak of the inflow hydrograph, which in the case of the Rustenberg wetland would amount to <1h, whereas the flow data were at a daily time step. Thus the approach used involved estimation of a regional non-dimensional flow duration curve (using all gauged unregulated similar-sized catchments in the area, standardized by long term mean discharge, and averaged), calculation of the actual flow duration curve at a site downstream of the wetland by multiplying the non-dimensional curve by the long-term mean discharge at the site, and conversion of the latter into a continuous stream flow hydrograph using the spatial interpolation technique.

The results suggested that the Rustenberg wetland does perform a flow regulating function by reducing the frequency of high flows and increasing the frequency of flows in the rest of the size range. This concurs with the findings of Woltemade and Potter (1994). Visual examination of the flood hydrographs with and without the wetland showed that flood peaks were reduced and lengthened (Smakhtin and Batchelor, 2005). In order to quantify the effect, Smakhtin and Batchelor (2005) compared the proportion of time that a high-flow or low-flow run is above or below an arbitrary threshold, with and without the wetland. This suggested a 23% reduction in the events above a threshold corresponding to the flow exceeded 5% of the time, and a 60% reduction in the events below a threshold corresponding to the flow exceeded 75% of the time.

Water held in the wetland and its soil, as well as water that infiltrates lower into the groundwater, contributes to base flows downstream. There is a base flow index (BFI) which provides an indication of river flow stability. Applied to the Rustenberg wetland, the BFI of 0.35 and 0.48 with and without the wetland suggested that 48% of flow was from subsurface stores with the wetland compared with 35% without the wetland. The 13% increase in base flow was attributed to the wetland (Smakhtin and Batchelor, 2005).

Apart from potentially contributing to downstream base flows as described above, infiltration within wetlands may also recharge aquifers that are tapped for groundwater use elsewhere. Estimation of this function also requires a comprehensive hydrological model, and there are extremely few examples of the quantification of this function (e.g. Thompson and Hollis, 1995). In the case of the Okavango Delta, the groundwater recharge was quantified using hydrological models of the delta (Jacobsen *et al., 2*005). Total groundwater abstraction around the delta was estimated to be 5.8 mm^3 per annum, which amounted to around 5% of the estimated annual recharge (Plantec *et al., 2*006; Turpie *et al., 2*006b). Because the latter value was smaller, and because the entire population living around the delta is reliant on the same aquifer for water, via boreholes and wells, this was the value required for estimation of the value of groundwater recharge.

Contrary to the above examples, Carter *et al.* (1979) suggest that downstream flows may be reduced by wetlands where the growing season coincides with the dry season, in which case the available ground water may be reduced through transpiration by wetland plants. Similarly, Novitzki (1979) claims that seasonal variances will be magnified by the presence of wetlands in a catchment. His findings concurred with Carter's, but excessive flow in the wet season was linked to the specific climate of the area of study (Wisconsin, U.S.A) where there is significant vegetation die back in the extremely cold winter months. There may be similarities in South Africa in areas where there is significant frosting in the winter (Kotze *et al.* (2008). These would be the higher lying areas that are susceptible to frost.

Whether a wetland performs a recharge function also depends on its location. Generally, a wetland above the saturation zone will act as a recharger, while those in contact with the main groundwater zone will serve as aquifer through-flow or discharge areas. The relationship between groundwater recharge and discharge may also vary intra- and inter-annually (Kotze and Breen, 1994).

Krasnostein and Oldham (2004) identified two components in the groundwater flow of the Loch McNess System in Western Australia. One was a regional one which was orientated

downhill towards the coast. The other was a local one which flowed according to the potential head difference between the wetland the surrounding groundwater. That is, in the dry summer, the wetland was recharged by the surrounding groundwater. When there was enough inflow through catchment run-off and the head between wetland was reversed, the groundwater recharge occurred.

The position of the wetland in the landscape is likely to play a significant role in determining the degree to which it performs a flow regulating service. Conger (1971, cited in Novitzki, 1979) states that 50% of flood attenuation will occur in the first 5% of a catchment's wetlands. This is due to each wetland's unique hydrology, the initial wetlands will each slow precipitation differently and so a flood peak after a high rainfall event will desynchronize. Wetlands that can store at least 25% of the catchment runoff from a 24-hour two-year rain event have a high water storage function (Forbes and Doyle, 2007).

4.4.3   Sediment retention and soil fertility

The sediment load carried by rivers is generally created by detachment of soil particles within the catchment area and is primarily due to precipitation and delivery of sediment to streams via overland flow (Vellidis *et al.,* 2003). When flows enter wetlands, they are slowed down and part of the load settles out in the wetland. This enriches the productivity of the wetland and also the agricultural potential of floodplains, a service that would be lost if the wetland was degraded.

In many cases, sediment yield from the catchment is accelerated by land disturbance, elevating the sediment loads carried by rivers. Wetlands can trap some of these extra sediments, thus reducing the potential damage caused by elevated sediment loads downstream. These damages would include the costs associated with increased turbidity of aquatic systems, siltation of aquatic habitats and siltation of water supply infrastructure and monitoring weirs. Higher silt loads in rivers may decrease light penetration and thus primary productivity, which in turn affects fisheries. Silt deposition within rivers decreases habitat and hence biodiversity in these systems. Siltation of dams and weirs reduces their capacity and lifespan, incurring costs through increased maintenance and/or augmentation schemes.

The ability of wetlands to remove excess sediment loads is related to their ability to reduce water velocity, and is thus closely related to a wetlands flow regulation capacity. Slope of the wetland is obviously a key factor (Novitzki, 1979, Boto, 1979), as well as the roughness and holding capacity of the wetland. As the water slows down, the energy required to keep

sediments in suspension is lost, and deposition occurs (Craft & Casey, 2000; Vellidis *et al., 2*003). Soil properties of the catchment area are also important, because the size of soil particles affects the degree of slowing required to induce settling (Boto, 1979).

There are a few examples of the rates of sediment trapping in the literature. For example, the Jackson Creek wetland – a 95 acre shallow prairie marsh containing three sediment retention ponds trapped 127-980 tons of sediment per year over three years (Elder and Goddard, 1996). The Old Woman Creek estuary, a coastal wetland in Ohio, traps 47% of incoming suspended sediment and has a sedimentation rate of about 1 cm/yr (Wilson *et al., 2*005). A study of 25 natural wetlands in the Washington D.C. area found an average removal rate of 75% of total suspended solids with some wetlands removing up to 93% (Anon., 1993).

Floodplain wetlands are frequently favoured as sites for agriculture because of the relatively high moisture and fertility compared with surrounding upland areas. In the Okavango Delta, for example, floodplain recession agriculture was approximately 40% more productive than dryland agriculture (Turpie *et al.,* 2006b). There have been numerous examples of a loss of floodplain fertility following upstream impoundments, such as the Aswan Dam on the Nile, which necessitated the introduction of irrigation and fertilisers.

4.4.4   *Waste treatment (water quality amelioration)*

Aquatic systems can play an important role in the absorption and breakdown of organic and inorganic pollutants. Organic pollutants, such as nitrates and phosphates, and inorganic pollutants, such as heavy metals, are diluted, taken up by plants, trapped along with sediments or broken down within aquatic systems. Water quality amelioration services obviously only occur downstream of where wastes are generated. This ability of wetlands to perform the service is thought to be related to ecosystem health, which is in turn related to the in-stream flow and management practices. Water quality amelioration services are of value wherever downstream water and ecosystem services occur that would be impacted by a loss of water quality.

The ability of wetlands to remove wastes has been recognized since the 1970s, resulting in the proliferation of man-made wastewater treatment wetlands and the use of natural wetlands for wastewater treatment. These wetlands are used to remove suspended solids, dissolved nutrients, pathogens and heavy metals, often providing an additional (tertiary) level of treatment to primary and secondary treatment processes (Ewel, 1997). Wastewater

treatment in South Africa still reflects a history of a biologically-oriented treatment philosophy compared to other regions of the world (e.g. Europe), where chemical treatment often took precedence. Biological treatment uses bacteria and micro-organisms to break toxicants and convert them for metabolism and other processes. Therefore, biological sewage treatment is superficially similar to natural treatment by wetlands (and artificial wetlands) except treatment works are much more concentrated.

Valuation studies that put a monetary value on this service frequently make the assumption that wetlands remove the total pollution input load. Accurate valuation of this service requires a much better understanding of the removal rates of natural wetlands.

A number of studies have been carried out on the water quality amelioration function in natural and created aquatic habitats (e.g. Jordan *et al., 2*003; Peltier *et al., 2*003, Thullen *et al., 2*005; Batty *et al., 2*005), but most research has been carried out in treatment wetlands. In South Africa there are data on the capacity of artificial wetlands to treat wastewater (e.g. one ha wetland can treat about 272 m^3 of wastewater per day – Rogers *et al., 1*985), but little data exists on natural systems, which are generally less efficient (A. Batchelor, Wetlands Consulting, pers. comm. 2007). However, while removal rates in natural wetlands may be lower than in man-made wetlands, they are often more permanent, because of the loss to peat. Most constructed treatment wetlands have to be reset every few years by removing the accumulated biomass, and often soil as well, as they become saturated (J.A. Day, University of Cape Town, pers. comm. 2008).

In treatment wetlands, absolute removal rates are often proportional to concentration, and the percentage of N and P influx removed tends to increase as the hydraulic loading rate decreases and the detention time increases (Jordan *et al., 2*003). Hydraulic efficiency, which is the degree to which a wetland disperses inflow over its area, is also important. This maximizes contact area and it can be assumed that it serves to increase detention time as well. Wetlands are thought to be better at removing total suspended solids, phosphorus and ammonia during high flow periods (when sediment loads entering the wetland increase), but better at removing nitrates during low flow periods (Johnston *et al., 1*990).

The removal of **phosphorus** is extremely variable. Waste water treatment systems usually report in the order of 30-50% removal, natural wetlands have removal efficiencies ranging from as low as 9% to as much as 85% (Anon., 1993). In treatment wetlands, some phosphorus gets incorporated into the biomass, but the excess is difficult to treat because it must be precipitated out using chemicals which increases the salinity of the water. Also, the

system may lose phosphorus as it becomes inert and cannot be treated further. Additionally, the extracted phosphorus necessitates careful disposal in landfills to prevent leaching.

Because phosphate is associated with sediments (Brinson, 2000), much of the load may enter the wetland during large flood events (McKee *et al., 2*000). These loads often become a permanent part of the bottom sediments, and wetlands with clay soils are particularly efficient at retaining phosphorus, even at low loading rates. Thus phosphate removal is expected to be higher in wetlands with low water velocities and high hydraulic roughness. Macrophytes also contribute to total phosphorus retention by enhancing sedimentation. High vegetative productivity and long water retention times also allow long-term phosphorus storage through the accumulation of litter and peat (Mitsch and Gosselink, 2000).

Phosphorus may be taken up in the wetland by plankton and periphyton, but this storage pool is small with rapid turnover (Richardson, 1985). Macrophytic production may account for measurable phosphorus uptake, however approximately 30-75% of the nutrient is seasonally released back to the water column during senescence, with some permanent storage as peat and litter (Richardson and Craft, 1993). Adsorption of dissolved phosphorus to soil and sediments is the main retention processes in wetlands. Sediment-litter is the ultimate sink for phosphorus (Faulkner and Richardson, 1989). Note, however, that at high phosphorus loading rates (e.g. wastewater effluent at concentrations of 2ppm or higher); wetlands may eventually become a phosphorus source rather than a sink (Tilton and Kadlec, 1979; Forbes *et al., 2*004).

Nitrogen is removed in wetlands by the nitrification-enitrification process (Saunders and Kalff, 2001). Nitrification is the microbially-mediated oxidation of ammonium (NH_4) to nitrite (NO_2) and then nitrate (NO_3). This process consumes oxygen and thus occurs in aerobic areas of the wetland. Nitrate then diffuses to anaerobic areas of the wetland where it may be denitrified. This is the rate limiting step in the removal of nitrogen from flooded systems.

In the denitrification process nitrate (NO_3) is reduced to gaseous nitrous oxide (N_2O) and nitrogen gas (N_2), which are then released to the atmosphere (Mitsch and Gosselink, 1993). This occurs mainly in sediments with abundant organic matter that provides a carbon source for denitrifying bacteria.

Nitrogen removal by treatment wetlands tends to be proportional to loading rate, with higher efficiencies or up to 75-95% at higher loading rates (Anon., 1993). In natural wetlands in the Washington D.C. area, removal efficiencies ranged from 5-60%, or 25% on average (Anon.,

1993). Numerous studies in the Northern Hemisphere have shown that a large proportion of the nitrate in subsurface flows moving towards streams was removed from the water as it passed through riparian areas.

Accuracy of estimation of the value of the wastewater treatment by aquatic habitats will depend on finding out absolute rates (e.g. g of N per year) of waste removal. Based on a review of available data, Verhoeven *et al.* (2006) provide a synopsis of current understanding of the removal rates per unit area (Table 4.3).

Table 4.3: Loading rates of wetlands in agricultural catchments in relation to relevant loading thresholds (Source: Verhoeven *et al., 2*006)

Catchment	Location	Wetland Type	Origin	N load (kg/ha/year)	P load (kg/ha/year)
Liuchahe	China	Multi-pond	Constructed	>500	>50
Legge, Twente	Netherlands	Riparian	Natural	200-1140	
Everglades	U.S.A.	Marsh	Natural	No Data	2-40
Mississippi	U.S.A.	Forested	Natural	19-39	2-9
Various	U.S.A.	Riparian	Natural	20-155	No Data
Treatment Wetlands (US, EU)			Constructed	500-9000	100-2000
Max load*				1000	60
Critical Load**				25	10

*Beyond this limit, the wetlands will show substantial leaching and associated high concentrations in the outflow
**Beyond this limit, wetlands with a species-rich vegetation will show a dramatic increase of productivity and associated change in species composition

Much of the variability reported from Northern Hemisphere wetlands may be related to the fact that treatment wetlands do not work under conditions of extreme cold. One may thus expect less variation in South African wetlands (J.A. Day, University of Cape Town, pers. comm. 2008).

Inputs with low nitrate loads (e.g. from natural catchments) will require both aerobic and anaerobic conditions to nitrify and denitrify nitrogen inputs. In this situation, nitrification is enhanced in wetlands where soil moisture contents fluctuate repeatedly (Patrick and Mahapatra, 1968, Ponnamperuma, 1972). On the other hand, wetlands receiving fertilizer runoff or other sources associated with high nitrate levels will reduce nitrogen loads most efficiently when they are anaerobic and when input nitrate concentrations are high. Wetlands that are more or less permanently inundated promote reducing conditions.

Landscape processes also need to be taken into account. Because of the common perception that wetlands act as pollution filters in a catchment, some authors have likened wetlands to a point source equivalent in a landscape dominated by non-point source pollution. However, waste uptake does not only occur within aquatic ecosystems, but also occurs during the drainage process, as waste-water runs through various habitats *en route* to streams and rivers. In Florida it was estimated that 9.3% of total nitrogen inputs of a catchment reached surface water and 19.6% reached the groundwater, the remainder being taken up in soils (Young *et al., 2*008).

4.4.5 *Ecological regulation*

Some ecosystems support organisms that help to keep pests under control. While this may be true of some aquatic ecosystems (e.g. fish that eat disease vectors), another important aspect is that aquatic ecosystem degradation can improve conditions for certain pests (e.g. reduction in flows leading to stagnant water ideal for mosquitoes and bilharzia, or invasive plants such as water hyacinth; Turpie and Van Zyl, 2002). Changes in flow might also affect the abundance or range of alien invasive fish species.

Artificial wetlands have also been shown to be efficient in the removal of bacteria and viruses (Gersberg *et al., 1*987).

The proliferation of pests, disease or invasive organisms has impacts on biodiversity and ecosystem health, and hence the output of ecological goods and services, and also on human and livestock health. In many cases considerable expenditure is made in order to control these organisms in order to prevent such damages. The expenditure on this control can be seen as a proxy for the potential damages if left uncontrolled. Alternatively, if no management measures are taken, the cost of these impacts are measured in terms of the change in value of aquatic ecosystems, plus the impacts on human and livestock productivity.

4.5 Cultural services

Recreational, spiritual, cultural, educational and other values that are loosely clustered under cultural services are derived from the attributes of wetlands. For example, recreational value might be attached to the presence of rare species, from the fact that wetlands add to landscape beauty, or their potential for angling. Spiritual value might be gained from

landscape beauty. Religious and cultural values might be gained from the provision of a place for conducting ceremonies. Educational and scientific value might be gained from the presence of un-impacted environments which provide an opportunity for understanding natural biological processes. In all of these cases, the estimation of current value does not require any quantitative biophysical information on the wetland. However, estimation of a change in these values due to a change in wetland condition would require estimates of changes in the parameters that affect these values. As is the case for provisioning services, these changes have seldom been quantified on the basis of statistically-accurate predictive relationships. The current tendency is for the estimation of change in the relevant parameters on the basis of expert opinion, and more recently, for the construction of response curves which are more explicit in the assumptions made.

4.6   Conclusion

In general, ecosystem regulating services are chiefly a function of the water detention abilities of wetlands, which enhances water regulation, vegetative productivity and sediment retention. The water detention ability is likely to be influenced by a number of factors:

1. Wetland size in relation to catchment area and runoff.
2. Wetland shape
3. Slope, which influences flow speed and water retention time
4. Holding capacity, which is influenced by soil type
5. Channel morphology, which influences water retention time. Hydrological modifications or modifications to wetland outlets typically reduce their effective storage volumes.
6. Roughness, which is a measure of friction and which can be attributed to actual soil roughness, the presence of depressions in the wetland, or resistance caused by vegetation.

The delivery of ecosystem services will also be related to ecosystem health, inasmuch as this affects the parameters relating to ecosystem delivery. For example, erosion of a channel may reduce the water detention capacity of a wetland. Thus it is necessary take ecosystem health into account in assessing the potential capacity of a wetland.

5. TYPES OF VALUES GENERATED BY WETLAND SERVICES

5.1 The concept of economic value

Economic value can be defined as the most that a person is willing to give up in other goods and services in order to obtain a good, service, or state of the world. In a market economy, money is a universally accepted measure of economic value, because the amount that someone is willing to pay for something tells how much of all other goods and services they are willing to give up to get that item. Thus their willingness to pay reflects the economic value. Market prices do not always accurately reflect economic value, since many people are actually willing to pay more than the market price.

The net economic benefit to individuals can be measured by "consumer surplus", which is the difference between total willingness to pay and the total amount actually paid. For society as a whole, this is measured as the area under the demand curve for a good, above its price (Figure 5.1). This value changes if the price or quality of the good or service changes, or if demand changes due to changes in prices or quality of substitutes or complements. Similarly, net benefits to firms or producers can be measured as "producer surplus", which is the revenue that they receive over and above the amount they were willing to accept for goods (to break even). For an industry as a whole, this is measured as the area above the supply curve and below the market price (Figure 5.1).

The total net economic benefit or cost of a change in an ecosystem is therefore the sum of consumer surplus and producer surplus, less any costs associated with the policy or initiative.
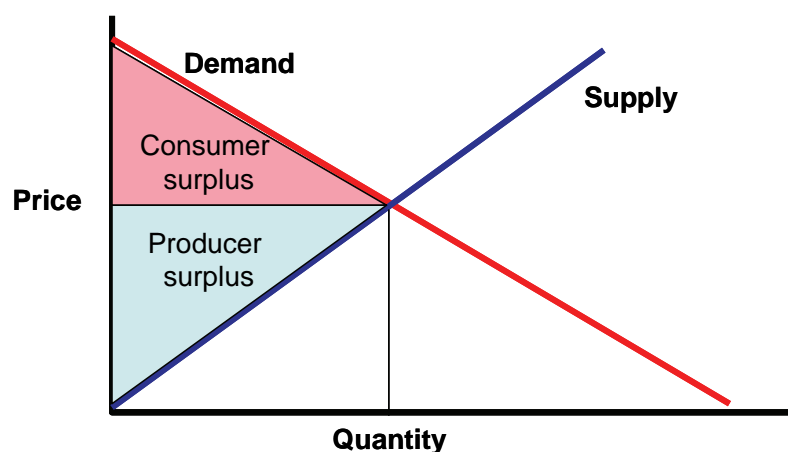
Figure 5.1: Demand and supply curves for a good, showing the calculation of consumer and producer surplus.

5.2 The Total Economic Value framework

Ecosystem valuation has generally been undertaken within the framework of Total Economic Value, which includes direct use, indirect use and non-use values. The total economic value generated by a wetland can be categorised into different types of value (Figure 5.2), providing a useful framework for analysis.

5.2.1 Direct use values

Direct use values result from economic activity and are generated through the consumptive or non-consumptive use of a wetland's natural resources. Direct use values are generated through crop production, livestock grazing, fishing, wild plant use and hunting (based on wetland goods). They are also generated through consumptive (hunting) and non-consumptive (wildlife viewing) tourism (based on wetland attributes). Wetlands offer a number of recreational activities such as waterfowl hunting, salt or freshwater fishing, game viewing, nature study opportunities and photographic subject matter (Bergstrom and Stoll, 1993). Direct use values also include aesthetic, spiritual and religious appreciation or use of wetlands (based on wetland attributes).
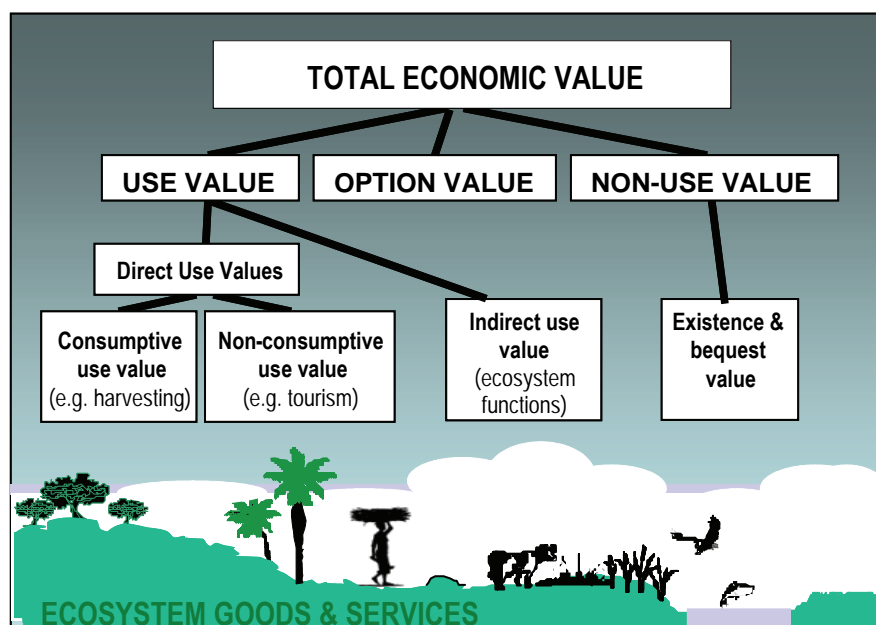
Figure 5.2: The classification of ecosystem values that make up Total Economic Value (based on Turpie *et al., 1*999).

5.2.2 Indirect use values

Ecosystem functions may either generate outputs that form inputs into production processes elsewhere (in other words the benefits are realised off-site), or they result in engineering cost savings by performing functions that would otherwise require costly infrastructure or man-made processes. These are the services provided by wetlands. Their value is generally positively related to the level of health or integrity of wetland systems. In this category one should also consider the disservices provided by wetlands, such as breeding grounds for pests and pathogens. These negative externalities often increase with wetland degradation.

5.2.3 Option value

Option value is the estimated future value of resources and services offered by the wetland such as possible medicinal, leisure, agricultural or industrial uses (Danielson and Leitch, 1986). Option value is particularly important when there is still uncertainty regarding the potential use and value of the wetland later on (Nhuan *et al., 2*003; Perman *et al., 1*996; Barbier, 1993). Even though a wetland may be underutilised at present, it may possibly be valuable for scientific research, education, tourism and other commercial enterprises which would increase its economic value in the future (Barbier, 1993). Another way that the option value of a wetland may be raised is if the local community have uncertain incomes (Nhuan *et al., 2*003). If there is no readily-available social welfare scheme for them to fall back on, they may be dependent upon basic commodities that they could harvest from the wetlands to tide them over. While option value cannot be measured, it is possible to estimate "quasi-option value", which is society's willingness to pay to retain the options for future use of the wetland (Perman *et al., 1*996).

5.2.4 Existence value

Existence value is the value of simply knowing that the resources or biodiversity within the wetland are protected. This value is sometimes divided into existence and bequest value, the latter being the value the current generation puts on the wetland area in order to preserve it for future generations (Pearce and Turner, 1990; Barbier, 1993). This is despite the fact that they do not derive any direct personal benefit. Local communities may regard the wetland as part of their heritage and link it to aspects of their beliefs, culture and traditions and therefore wish to be able to pass their customs and heritage that have developed around the wetland onto their future generations (Barbier, 1993). It is extremely difficult however to predict future option values as these will be closely correlated to future

incomes and people's preferences (Barbier, 1993). Although far less tangible than the use values, non-use values are reflected in society's willingness to pay to conserve these resources, and with appropriate market mechanisms, can be captured through transfers and converted to income.

5.2.5   Total economic value

Total Economic Value (TEV) is theoretically the sum of all the above values, although depending on how they are measured they may not always be additive.   The main consideration in adding values is to make sure that there is no double-counting.

Direct and indirect use values are of particular importance in a developing country context, for which a critical national objective is to create growth in income and employment.   These values are manifested directly or indirectly in tangible income and employment.   Existence values inherently are not manifested in income and employment, and they are often highest in developed countries.   Nevertheless, global existence values can be high and the resultant willingness-to-pay can be captured globally and converted to national income, for example through grants.

5.2.6   Aligning the TEV framework to concepts of ecosystem services

The TEV framework aligns directly to either the goods, services and attributes framework, or the Millennium Assessment concept of ecosystem services, apart from supporting services, which underlie the services that are valued (Table 5.1).   A subtle difference between the ecosystem services concepts is that recreational fishing or hunting would be classed as a consumptive use under the traditional framework, but as a cultural service (rather than a provisioning service) under the Millennium Assessment framework.

Table 5.1:   The way in which the original and the Millennium Assessment concepts of ecosystem services relate to one another and to the components of Total Economic Value

Goods and Services	Millennium assessment	Total Economic value
Goods	Provisioning services	Consumptive use value
Services	Regulating services	Indirect use value
Attributes	Cultural services	Non-consumptive use value
		Option value
		Existence value
n/a	Supporting services	n/a

6.   VALUATION METHODS

6.1   General approach to valuation

In endeavouring to evaluate the economic value of wetlands, a researcher must aim to not only employ the most scientifically sound methods and analyses he can, but at the same time needs to realistically fit these within the limitations of a set budget, a specific time-frame, available skills such as interpretation and data capturing, and the data that are feasibly obtainable (Nhuan *et al., 2*003). In general, wetland valuation studies should follow the guidelines set down by Barbier (1994), Nhuan *et al.* (2003) and De Groot *et al.* (2006).

Barbier (1994) introduced the following framework to valuation studies:
1. Choose an appropriate general assessment approach within which to apply valuation methods.
2. Define the scope and limits of the valuation and information needs:
    - geographic and analytical boundaries,
    - time frame,
    - identify the basic characteristics of the area in terms of structural components and functions, and also attributes, e.g. biodiversity, cultural uniqueness,
    - determine the type of value associated with each, e.g. direct consumptive use value
    - rank the major characteristics and values, e.g. in terms of relevance to the study, or contribution to overall value, and
    - tackle the most important values first, and the least important only if it becomes necessary.
3. Define data collection methods and valuation techniques.

Nhuan *et al.* (2003) suggest the following simple steps be taken when approaching wetland evaluation:
1. Appropriate evaluation methods need to be decided upon, which are suitable for the particular research objectives being proposed. For developing national conservation strategies a total economic evaluation is advocated.
2. Delineate the boundaries of the wetland area as accurately as possible. This may require the consultation of maps which give the required information on soil types, vegetation zones, flood lines and agricultural practices.
3. Find out what the key resources and assets offered by the wetland are and make a list, ranking them in terms of their priority. This information may be obtained from previous

literature written in the form of scientific papers, consultancy reports and national resource inventories.

4. Investigate whether each of the different functions and services offered by the wetland has a direct, indirect or non- use benefit associated with it.

5. Identify the types of information required to evaluate each category of use value being investigated and plan how to source this data.

6. Estimate the wetland's economic value.

7. Implement an appropriate appraisal method, such as cost-benefit analysis (CBA) or multi-criteria decision-making. This choice will affect all of the seven steps in the approach to evaluating the wetland (Nhuan *et al., 2*003).

Finally, De Groot *et al.* (2006) advocate the following steps for undertaking valuation as follows:

1. Policy analysis (why value?);

2. Stakeholder analysis (who does it and for whom?);

3. Function analysis (what should be valued?);

4. Valuation of services (how to value?); and

5. Communicating wetland values (to whom to provide the results?).

Any of these approaches is valid, the first two emphasising technical approaches, and the third emphasising the need for stakeholder buy-in. The message they provide in common is to undertake sufficient prior assessment of the situation to allow appropriate and effective use of the available valuation methods in order to meet the research requirements.

6.2   Types of valuation methods

For many wetland products there are markets, and it is relatively easy to estimate their worth (Barbier *et al., 1*997). However, in many instances, prices are distorted and may not necessarily reflect the social value of resources (Perman *et al., 1*996). It also is far less simple to value the biodiversity within wetlands or to quantify how valuable the aesthetic aspect of a wetland's landscape is to society.

The methods used to value economic goods and services of wetlands are no different from the methods used to value any other type of environmental assets. Different types of value are each measured with a different choice of methods (Table 6.1). The number of possible methods that can be used to measure the different types of values also decreases from left

to right along the columns in Table 6.1. Option value is seldom measured explicitly and is also fairly difficult to separate in practice from existence value.

Valuation methods can be divided into three main categories: market-value approaches, surrogate-market approaches and simulated market approaches. While surrogate- and simulated-market approaches can measure the demand for wetlands, and hence willingness to pay, market value approaches are based on market prices (revealed willingness to pay) and do not necessarily include consumers' surplus, or peoples' willingness to pay over and above what they actually have to pay (see Turner *et al., 1*997). The latter group of techniques thus normally underestimate benefits.

Each of the most commonly used methods is discussed below. The published literature tends to pigeonhole environmental valuation techniques into discrete methods. However, it is important to note that many of the 'methods' mentioned in the literature do not stand alone as valuation techniques, but form part of an overall approach, often involving the combination of different methods, and innovation, where appropriate. The simpler methods produce a total value, whereas those that involve construction of models are better for estimating marginal values (the additional value generated by each unit of production).

Table 6.1: Commonly-used natural resource valuation methods, and the types of value which they are generally used to measure (XX = main use, X = possible use)

	Direct use values		Indirect use values	Option and non-use value
	Consumptive	Non-consumptive		
Market value approaches				
Market Valuation	XX	X		
Production Function	XX	X		
Replacement Costs / Avoided Damage etc	X	X	XX	
Surrogate market /revealed preference approaches				
Travel Cost Method	X	XX		
Hedonic Pricing	X	XX	XX	
Simulated market /stated preference approaches				
Contingent Valuation Methods	XX	XX	X	XX
Conjoint Valuation	X	X	X	X

6.3   Market value approaches

These methods are commonly applied to the measurement of both direct and indirect use values of natural systems, and can similarly be applied to the valuation of wetland areas. These methods can be applied to direct or indirect use values when the wetland provides an unpriced input into any 'production process', for example, the view of a natural wetland as an input into a tourism venture, or the use of a river as a conduit for stormwater – i.e. an engineering process.

6.3.1   Market valuation

Market valuation uses standard economic methods to value goods or services that are bought and sold in the market place. The exact types of costs and prices used depends on how the value will be expressed, e.g. economic surplus, net private income, gross economic output or direct value added.

Estimation of economic surplus involves estimation of consumer and producer surplus, using data on quantities bought and sold at different prices. Consumer surplus is the maximum amount that people are willing to pay minus the amount that was actually paid. Producer surplus is the revenue minus the variable costs of production. For example, if one was measuring the value of a fishery, it should involve (1) estimating a demand curve for fish and working out the consumer surplus, then (2) estimating the total revenues of the fishers and subtracting the variable costs to estimated producer surplus.

Many valuation studies of direct use activities (e.g. grazing, fishing, tourism) concentrate on the production aspect (ignoring consumer surplus); estimating the producer surplus, net private income, or the value added to national income by the production activity. This approach provides an estimate of the contribution of the wetland to output value in the production process. If all the other inputs are priced, then the wetland value is estimated as the gross income from the final product minus the costs of the priced inputs. This is critical, since many valuation studies have failed to separate out the value that is contributed by other inputs.

6.3.1.1 The measurement of outputs, prices and costs

In the case of subsistence or commercial uses of wetland resources, this method would entail estimating quantities harvested or produced, market prices (or barter equivalents) and

input costs (e.g. nets). For tourism use, this might refer to number of bed nights sold, market price and the costs of running the tourist establishment.   The measurement of these quantities, prices and costs is not always straightforward, however.

Output levels, such as the quantity of resources harvested, can be estimated in a number of ways depending on the accuracy required. Although much attention is given in the literature to the measurement of value in valuation studies, comparatively little is said about the measurement of quantity. Quantities of outputs can be measured using existing data, e.g. fisheries catch statistics, by direct observation, such as obtaining observations of fish landings, or by survey instruments, where respondents are asked to recall their harvests. It may be possible to use statistics published by government offices and to conduct surveys in nearby markets (Nhuan *et al., 2*003).   Whereas time series data on resource use is often available in developing countries, these data seldom exist in a reliable form in developing country contexts (Eaton and Sarch, 1997, Emerton, 1998, Turpie *et al., 1*999).   This necessitates reliance on survey data, which means reliance on the recall ability of respondents. These instruments are discussed further below. In the case of commercial and illegal harvests it is often extremely difficult to elicit accurate information on production quantities. In all cases, it is important to take variability, such as seasonal variability, into account when quantifying resource use.

Where market **prices** for harvested resources are available, these should serve adequately as measures of value (Barbier *et al., 1*997, Batie and Shabman, 1982), unless price distortions are expected. The type of price used should be stated explicitly in valuation studies. Prices are often taken at the 'farm-gate' level, in other words price accepted by the harvester, before any value is added to the resource by marketing or processing. However, it may be more appropriate to consider the full value generated by a wetland area, right up to the final consumer or export.

If no market prices are available for a resource, as is often the case in subsistence economies, then surrogate prices can be used. There are several possible ways of doing this (Barbier *et al., 1*997):
1. *Barter or trade value*: If the resource is bartered or traded, e.g. fish for rice, then it may be possible to estimate its value based on the market value of a commodity for which it is traded.   This method requires information about the rate of exchange between two goods. If such trade is not observed the information can be obtained using properly-designed survey instruments, e.g. ranking techniques in a focus-group discussion.

2. *Substitute price*: If a close substitute can be identified which has a market value, then it is possible to assign the value as the price of the substitute. This requires information about the degree of substitution between different goods. It is also possible to estimate the amount of money people save through using natural products as opposed those bought in markets (Delang, 2006a).

3. *Opportunity cost*: Alternatively, it is possible to derive a minimum value for a good by estimating the opportunity cost, or value derived from the next best use, of the inputs (e.g. capital or labour) required for its harvest or production. However, this method may prove difficult when people do not harvest resources in a systematic way (e.g. people collecting materials on their way home from working in the field) (Delang, 2006a).

4. *Indirect substitute prices*: In the absence of all the above possibilities, and when the substitute is also unpriced, then it may be necessary to use the opportunity cost of the substitute as a proxy for the value of the commodity in question.

Where **inputs**, such as fishing nets or tourism lodges, are required, their costs can be estimated directly using market prices. Economic costs of inputs would include labour costs and the cost of raw materials. These inputs need to also be classified as either those which are paid for (e.g. tools and equipment, outside labour and licences for harvesting resources) or as free inputs (e.g. family labour and borrowed equipment; Nhuan *et al.,* 2003). For harvested resources and subsequently manufactured goods which are sold it is useful to record the producer's price, transport costs to markets and the final selling price.

In an economic analysis it may be necessary to adjust the above prices and costs by **shadow-pricing**, if market distortions are suspected. Under certain conditions, market prices may not reflect the true value of a resource or the true input costs. Prices may be distorted by conditions of imperfect competition, for example when local markets are relatively isolated, or through government intervention. If distortions are suspected, the use of shadow prices is usually advocated (Batie and Shabman, 1982, Barbier *et al., 1*997), but only if they can be adequately estimated (James, 1991). Shadow prices are corrected prices, to account for the distortions, and aim to reflect the full value of a commodity to society. They thus reflect economic value rather than financial value. However, the proper correction of distorted prices relies on accurate diagnosis of the direction and magnitude of the distortion, which is often difficult. For example, the costs of harvesting resources also includes labour time, which is usually taken as some proportion of the wage rate, or the shadow price of labour. Where opportunities for formal and informal employment are very low, the shadow price of labour time to collect natural resources approaches zero. This is a

complex issue, however, as all time could be said to have an opportunity cost in terms of other tasks or recreational activities that could have been carried out at that time.

6.3.1.2 Social survey methods

Where obtaining data production, prices and costs requires surveys of users, these surveys can take the form of key informant interviews, focus group discussions, or household questionnaire surveys, and may involve direct questioning and the use of various Rapid Rural Appraisal techniques (e.g. Turpie *et al., 1*999; 2006b). Although questionnaire surveys theoretically provide the most statistically rigorous quantitative data, there are many problems with such surveys that are better addressed by the more participatory techniques.

Key informant interviews are interviews with key members of the community that have broad or specialised knowledge on the use of resources, the tourism industry, etc. **Focus group discussions** involve a group of five or six people from the community. Discussions may be on various resource-harvesting activities such as fishing or hunting. In both cases, the discussions typically follow an unstructured set of questions but any additional information gleaned is not ignored. Focus group discussions and key informant interviews are held to collect information of a generally applicable nature, e.g. on seasonality, markets and prices, as well as to collect sufficient information to be able to make preliminary estimates of natural resources harvesting and processing and associated economic values, in order to assist with survey design. The discussions typically employ techniques developed under the banner of "participatory rural appraisal", such as developing resource maps and seasonal calendars. In a study in Nigeria local indigenous technical knowledge was an important component (Adaya *et al., 1*997). Local participatory researchers were invaluable as they were able to identify local communities' concerns about how they survive. Participatory research tools facilitate overcoming communication barriers and also help take into account the diverse needs of the community.

These types of surveys also tend to be long and complex and are therefore extremely reliant on good survey design and enumeration. **Household surveys** are used to collect quantitative data on natural resource use and other household activities (Turpie *et al., 1*999; Nhuan *et al., 2*003). These survey instruments are generally long and good design is critical to the accuracy of the study. In the case of large, heterogenous wetland areas, the area may be zoned into socially and ecologically similar areas (e.g. Turpie, 2000; Turpie *et al., 2*006b). These surveys establish the household composition, location and employment status, obtain details on each of the resources harvested, the equipment used, the amount

harvested annually, the quantity sold as raw produce and the selling price per unit, the number of products produced from natural products and the amount sold and the selling price of these. Data are also obtained on the areas of land cultivated, the type of crops grown and amounts harvested, as well as livestock numbers and production.

However, the accuracy of data diminishes when household surveys require quantitative information on past seasons (Delang, 2006a). Ideally the best way to quantify and record what is being harvested is to conduct regular surveys as the resources are being harvested and weigh and identify them. This would require conducting a study over at least a year however, and in most cases this is unfortunately not feasible.

6.3.2   Production function approach

The production function approach takes the above one step further in that it allows the estimation of marginal values, or change in value that will occur with a change in wetland area or quality (Ellis and Fisher, 1987, Barbier, 1994). The amount of a good or service (such as reeds, flowers, and tourism attractants) provided by an area is dependent on the qualities of that area, as well as the inputs (e.g. labour) involved in its production. For example, the harvest value of reeds from a wetland is a function of the depth and water quality of the wetland as well as of the labour inputs involved. This involves the estimation of a production function which has the wetland good or service as an input, follows:

$$Q = f(S, Xi…Xn)$$

where Q is the commodity produced by the stock of wetlands (S) along with other inputs, X (after Barbier, 1994).

It is important that the relationship between the wetland characteristics and the economic activity they contribute to is well understood. Ideally, this approach demands an understanding of the relationship between the output and the state of the environment, or the physical effects on production of changes in a wetland resource, and should be modelled taking dynamic functions into account. This is usually achieved through time-series or cross-sectional analysis, and thus usually requires data spanning a number of years or comparable data from a number of areas.

6.3.3   Restoration Cost or Replacement Cost methods

Some wetlands services can best be valued in terms of the costs that would be incurred if they were lost. This is especially useful in valuing ecological services such as the protective

function of wetlands. For example one might estimate the cost of building dams to replace a wetland's flood amelioration function. This uses the costs of restoring ecosystem goods or services (e.g. through habitat restoration), or of replacing them with artificial substitutes. Replacement costs are usually easier to estimate than restoration costs.

6.3.4   Damage Costs Avoided

This method estimates the cost of repairing the damage that would be incurred with reduction or loss of the wetlands area. For example, in estimating the flood protection value of a wetland, this would require the estimation of the costs (e.g. damage to houses, roads and other infrastructure) that would be incurred if floods were not ameliorated by the wetland. The method usually has to incorporate some type of probability analysis to estimate the probability and degree of damage that would occur. This method assumes that damage estimates are a measure of value, or in other words, that the damage is worth avoiding. Estimating marginal values requires an understanding of the relationships between wetland characteristics and their functioning.

6.3.5   Defensive Expenditure method

Instead of focusing on costs of 'repair', this approach focuses on the costs of prevention. This method uses the costs that would have to be incurred in preventing damage if the wetland was degraded or lost as a proxy of the value of those benefits. For example, the value of flood protection by wetlands could be measured as the cost that would be incurred in building dykes if a wetland was reclaimed.

6.4   Surrogate Market/ Revealed Preference approaches

6.4.1   Travel Cost Method (TCM)

This technique is used primarily for the valuation of recreational benefits of environmental amenities, especially in cases where visitor fees are low or non-existent (e.g. Clawson and Knetch, 1966; Willis and Garrod, 1991a; Bockstael, 1995; Turpie *et al., 2*001; Turpie and Joubert, 2001). The method derives willingness to pay for the use of an area from observed visitor behaviour. It is assumed that the money and time spent on visiting a recreational site is a proxy for the value of that site.

Information on travel costs is used to derive a demand curve for the site, from which the consumers' surplus can be calculated. Consumers' surplus is the extra amount over and above what they had to pay that people would have been willing to pay to use the area. Recreational value is comprised of total expenditure plus total consumers' surplus, although only the former contributes to the money economy. The demand function incorporates a number of factors which might influence peoples' visitation rates. It is possible to apply either a Zonal TCM or an Individual TCM. For a Zonal TCM, which is the preferred method, the visitors are divided into a number of origin zones based on travel distance from the recreational site, and the number of people coming from each origin zone and the average travel cost from each origin zone are calculated.

The data requirements for a travel-cost estimation are quite substantial and include the use of questionnaire surveys. Data requirements include the number of visitors to the site, their origin, socio-economic characteristics, the duration of the journey and time spent at the site, direct travel expenses, values placed on time by the respondent, and purposes of the journey other than visiting the site. The cost of visiting the site includes the opportunity cost of time, and the method of estimation of this opportunity cost is controversial (Smith *et al.,* 1983, Matthews, 1987, Shaw, 1992). The shadow price of time is often taken to be about a third of the wage rate, but it is not always considered appropriate to apply an opportunity cost to recreation time at all.

Visitation rates are described as a function of travel costs by fitting an appropriate model, which is usually a semi-log function in which the logged visitation rate is negatively correlated with the travel costs to the recreation site. This model is then used to determine the theoretical response of visitors to a range of prices of the environmental amenity, and the latter relationship is used to produce a hypothetical demand curve. Visitors' consumer surpluses are then calculated on the basis of this demand curve, using integration.

Although relatively straightforward in theory, the Travel Cost method is often complicated by several factors. The most confounding problem is that of journeys to multiple destinations, in which case it becomes difficult to isolate the value of the site in question from that of other destinations on the journey. Thus the method works well in situations where visitors to the site tend to be on single destination trips (e.g. Sandvlei – Turpie *et al.,* 2001). Another difficulty is accounting for substitute sites. A visitor may travel 20 km to visit a site which they particularly enjoy, whereas another who has less enthusiasm for the area may travel the same distance because there is no other available recreational site near home. The travel cost method would assume they have the same recreational value (or willingness to

pay) for the site. Some people may have low travel costs because they choose to live near the site in order to gain easier access. The travel cost method also is unable to account very well for the value held by people that walk or cycle to the site (JK Turpie, unpublished data), which is likely to occur very often for many urban wetlands areas, such as parks.

6.4.2 Hedonic Pricing Method

Through linear modelling of the different variables that contribute to property value, it is possible to calculate the contribution made by an environmental variable (such as a view of a wetland; Perman *et al., 1*996). This is because the value of environmental amenities is reflected in the prices paid for property, such that

$$P = f (E, X1, X2….Xn)$$

where E is the environmental variable and $X_1..X_n$ are variables such as size, aspect or other building attributes.

This method is inappropriate in situations where there is no market for property, such as in squatter areas, or where the markets for property are distorted (e.g. under the old group areas act). However it has been successfully applied in South Africa for valuation of estuaries (e.g. Turpie and Joubert, 2004; Turpie, 2006) and wetlands. Where resources are limiting, it is also possible to obtain reasonable estimates by means of estate agent interviews (Van Zyl and Leiman, 2002).

6.5 Simulated Market/ Stated-preference approaches

Simulated market (or 'Stated preference') methods provide the only means of estimating option and non-use values, and have also frequently been applied to the measurement of recreational use value. These methods are highly controversial, with some arguing that they should not be used at all (Desvouges *et al., 1*993; Diamond and Hausman, 1994; Carson *et al., 2*001). Stated preference methods should not be used to estimate the value of ecosystem services, since most people do not adequately understand these complex functions and their linkages to other economic activities.

6.5.1 Contingent Valuation Methods (CVM)

Contingent valuation methods elicit people's willingness to pay for access to or the existence of natural resources by means of direct questionnaire survey techniques (e.g. Mitchell and

Carson, 1989). In the survey, a hypothetical question (or set of questions) is posed to each of the respondents which elicit their willingness to pay for the preservation of biodiversity or their willingness to accept compensation for the loss of biodiversity. The method can be applied in a number of ways, and can approach the problem directly or indirectly. Willingness to pay can be elicited by means of open-ended questions, referendum or dichotomous choice (yes-no) type questions, bidding games, trade-off games, ranking techniques, costless-choice options, or the priority evaluator technique. The method usually has to be tailored to suit each unique valuation situation. As for travel cost surveys, contingent valuation surveys also obtain information on socio-economic factors, distance from site, etc in order to construct a demand curve from which the net social value can be estimated.

Because they rely on direct questioning rather than observing people's actual behaviour, contingent valuation methods are open to a number of biases. Indeed much of the academic literature on contingent valuation has paid attention to these biases and to finding ways of minimising them (e.g. Bishop and Heberlein, 1979, Willis and Garrod, 1991b, Cooper and Loomis, 1992, Carson *et al., 1*996). An important bias to be wary of is 'strategic bias' whereby respondents over- or understate their true willingness to pay because they believe their response may influence decision making. 'Embedding bias' occurs when people do not see the question in the context of all their wants, needs and budgetary constraints. 'Interviewer bias', 'information bias', 'starting point bias' and 'hypothetical bias' tend to steer the thinking of the respondents, and decisions are also influenced by the choice of payment vehicle (e.g. taxes or donations). Because of these biases and the difficulty in their resolution, the use of contingent valuation methods in valuation studies is somewhat controversial. This controversy led to the formation of a panel which examined the validity of the method and formulated guidelines as to its application (Arrow *et al., 1*993). Thus, if used properly in such a way as to minimise bias, it is deemed an acceptable method of measuring value (Arrow *et al., 1*993).

Among the recommendations of the American National Oceanic and Atmosphere Administration (NOAA) panel (Arrow *et al., 1*993), is that interviews should be face-to-face rather than telephonic or postal and they should be pre-tested. The valuation question should be in a willingness-to-pay rather than willingness-to-accept format where possible, and should be a referendum-type (yes-no) question, rather than open ended. However, when using a single referendum question, a sample size of at least 1000 respondents is required. Accurate information should be presented, and respondents should be reminded

to consider their own budgetary constraints and other expenditure preferences as well as the issue in question.

As with other valuation methods, contingent valuation methods have largely been developed in countries of the North. The application of contingent valuation methods in developing countries is somewhat more difficult and controversial than most methods.

6.5.2 Conjoint Valuation Methods (Choice Modelling; Contingent Ranking)

Conjoint valuation methods, also known as choice modelling or contingent ranking methods, were developed in the field of marketing, and are increasingly being applied to the valuation of environmental resources, including wetlands (Green and Rao, 1971; Stevens *et al., 2*000; Turpie and Joubert, 2001; Johnston *et al., 2*002; Carlsson *et al., 2*003; Hanley *et al., 2*006; Birol *et al., 2*006). These methods seek to ascertain the way in which different components of a 'package' of goods, or in this case, the different attributes of a wetland, contribute to its overall value, and the way in which this overall value changes when certain attributes change. For example, conjoint methods might be used to investigate the contribution of management efforts to the value of a wetland, and the impact of a change in management input on the overall value of the wetland. Or they may be used to investigate the effect of changing relative amounts of different types of wetlands. This is particularly useful for the analysis of multiple scenarios.

The technique involves questionnaire surveys in which respondents are asked to rank or score a range of scenarios which differ in the state of their attributes. There are two possible approaches: a two-factor evaluation or trade-off approach where only two attributes are considered at one time and a number of attribute combinations need to be considered by each respondent; or a multi-factor evaluation where respondents are presented with a combination of all attributes at one time. The latter is easier to administer, and will often reflect more realistic scenarios. Within multi-factor conjoint analysis, respondent burden grows exponentially with the number of attributes and attribute levels. For this reason, it is better to keep the numbers of these to a minimum while at the same time trying to ensure that a representative range of relevant wetlands attributes and levels is used.

Once the attributes and their levels are selected, a manageable number of feasible scenarios are selected (see Stewart *et al., 1*993). No respondent can be expected to evaluate more than 5 scenarios, so if 20 scenarios are used, then they would have to be

spread across at least 5 questionnaire versions, with one or two scenarios being common to all versions.

In a multifactor conjoint analysis, a single rating is given to a combination of attributes of different levels. Some measure of relative value of each scenario needs to be applied by the respondent. Binary preference models can be used (limited to 1's and 0's), and can be analysed using simple logit models, but these methods do not give very satisfactory results. A rating system such as score on a scale of 1 to 10 is usually used, in which ratings explicitly indicate the relative benefits associated with each scenario. These data can be analysed using an ordered probit model, or where the data are censored (such as hypothetical expenditure rankings), a tobit analysis may be better. They may also be analysed using a generalised linear version of multiple regression. The output may take the following form:

$$\text{Utility } (Z) = \text{constant } (K) + \beta 1 X 1 + \beta 2 X 2 + \beta 3 X 3 + \ldots,$$

where X_1 to X_n represent attributes of the area in question. In other words, utility (or value) is a function of the level of each of these attributes. Variables may be continuous (e.g. size in ha) or categorical (e.g. seepage wetland *vs*. floodplain wetland). One of the attributes may be monetary (e.g. entrance fee). In other cases, the value associated with different utility levels has to be estimated by including other valuation methods, such as contingent valuation (e.g. Turpie and Joubert, 2001).

6.5.3   Benefits Transfer

In certain cases it may be possible to apply the results of other studies of similar areas to the area under consideration (Georgiou *et al., 1*997, Barbier *et al., 1*997). This is called 'benefits transfer' because the measured benefits are 'transferred' from a site where a study has been carried out. It is then assumed that the existing or adjusted estimate of economic value can be used as an approximation of the economic value of the good or service in question.

There are three approaches to benefits transfer (OECD, 1994, Georgiou *et al., 1*997):

1. Transferring mean unit values
Here it is assumed that the wellbeing experienced due to an environmental good or service at one site is the same as the next. The problem is that at the new site, individuals may not have the same preferences.

2. Transferring adjusted unit values

The mean unit values obtained at a different site can be adjusted for any biases that are thought to exist, or in order to better reflect the conditions at the new site. Potential differences that should be considered are differences in socio-economic characteristics of individuals, differences in the environmental change being examined, and differences in the availability of substitute goods and services.

3. Transferring the demand function

Instead of transferring adjusted or unadjusted unit values, the entire demand function estimated at existing sites could be transferred to the new site. This is much better than the above methods, as the value would be altered depending on the specific characteristics of the new site.

There are two main advantages of the benefits transfer approach: firstly, economic benefits can be obtained more quickly than by undertaking primary research, and secondly, it is considerably cheaper. However, extreme caution should be applied in resorting to this 'technique', and unless very well justified, the resultant values should not form the sole basis of any major decisions. Studies that have tested the reliability of the benefits transfer approach (e.g. Shrestha and Loomis, 2001; Barton, 2002; Rozan, 2004; Hanley *et al., 2*006) largely reject the validity of this approach, due to the high degree of dissimilarity in both mean values and value functions between sites.

7.   PUTTING VALUES INTO PERSPECTIVE

7.1   Who requires what kind of values for what decisions?

While the Total Economic Value framework defines the types of wetland values to be quantified, there are various ways in which these values can be expressed, depending on who requires the information and for what decision. Values can be measured at a local, regional or national level, and from a private (financial) or a social (economic) perspective.

Different measures of value are relevant to different decision-makers. Individuals and firms may make decisions on the basis of their own financial and/or utility gains, whereas governments would tend to be more concerned with overall welfare gains (e.g. contribution to national income and employment). At a more local government level, municipalities may make decisions based on the generation of revenues, e.g. from property rates. It is

important to understand value both from an individual/firm perspective and a government perspective, since the former constitute the market forces of change, and the latter are required to make decisions that are in the overall interest of society.

7.2   Current value

In most decision-making contexts, such as water resource allocation, conservation and development planning, project appraisal and the like, it is appropriate to express wetland values in terms of the net economic benefits they generate, as described in section 3 above. This will usually include estimates based on total willingness to pay in the case of intangible values such as existence value. Net economic value is usually expressed as an annual value, and is the most common way of expressing the value of an ecosystem.

7.3   Net Present Value (NPV), discounting and sustainability

7.3.1   Estimation of net present value

In many cases it is also very useful to consider the value of the wetland over a period into the future, especially where sustainability is an issue. This is important for the evaluation of project or management alternatives, as well as for natural resource accounting (see below). Expressing the value of a wetland as an asset requires computing its net present value, which reflects the flows of values that it generates over time in a once-off price, just as the price of a farm reflects the expected future income from that farm. The value of a wetland area is thus the net present value of the flow of goods and services from the present until some specified time in the future. In calculating net present value, two decisions have to be made:

1. the **time frame** of the analysis; and
2. the relative weighting of future and present values, determined by the choice of **discount rate**.

The time frame of analyses is usually in the region of 10 to 50 years. While longer time frames are of more interest to ecologists, shorter time frames are more commonly used because the lifespan of policy is usually relatively short, and because of the effect of discounting on future values. Under most circumstances, values accruing beyond 20 years into the future are rendered negligible in present terms by discounting, and so 20 or 30 years is a common time frame for analysis.

Discounting converts future values to their equivalent value in present terms. The rate of discounting determines how future values are weighted relative to present values, with a high discount rate having the effect of down-weighting values accruing in the future. The net present value of future net benefits is determined by the discount rate as follows:

$$NPV = \sum_{t=0}^{T} \frac{(B_t - C_t)}{(1 + r)^t}$$

where B_t and C_t are benefits and costs at time t and r is the rate of discount.

A zero discount rate would give equal weighting to present and future values. With zero discounting, the net present value of a stream of net benefits would be equal to the sum of the net benefits in each year into the future. High rates of discounting, on the other hand, mean that future values are not worth much at all, and values accruing in the near future are far more important than those accruing in the distant future.

7.3.2   *The choice of discount rate*

The rate of discount reflects the investor's or society's rate of time preference. For a private owner or investor, this is influenced by the rate of interest that they could obtain on their investments. For example, if capital grows at a real interest rate of 10%, then in theory, the investor should be indifferent between receiving an amount of R100 in the present or R110 in a year's time. Similarly, the present value of a next year's earnings of R110 will be R100, calculated by applying a discount rate of 10%. The discount rate can thus be based on the real rate of earning interest on investment accounts or the interest costs of borrowing capital. Discount rates based on these interest rates can be considered to be 'private' discount rates in that they reflect individual rates of time preference. In reality, private discount rates will be higher than this when the risk of poverty, starvation or death is high.

However, in the case of publicly-owned wetlands or projects, it is more appropriate to use a social rate of discount. Indeed, the conservationist argument is that a low or social discount rate should be applied when valuing environmental costs and benefits in general. Social discount rates are usually lower than private rates, because society as a whole places greater value benefits and costs to future generations than individuals do. While there is no interest rate proxy for a social discount rate, two possible options exist: the Ramsey rule and hyperbolic discounting. The Ramsey rule is that the discount rate should be the sum of (1) the pure rate of time preference and (2) the growth rate of income multiplied by the elasticity of the marginal utility for money. The first component implies discounting of future utility *per*

se, while the second implies discounting the value of future consumption goods based on the nation that we will be richer in the future and that the rich gain less welfare than the poor from a given quantity of money (Sterner, 2007). Hyperbolic discounting, which uses declining rates over time, is consistent with Ramsey discounting.

7.3.3   Impact of unsustainable use on wetland value

Most valuation studies implicitly assume that the resources are used both sustainably and optimally. In converting annual benefit streams to net present values, it is simplest to assume that the magnitude of the benefit stream will be maintained for the duration of the time frame of the analysis. This carries the implicit assumption that resource use and other productive activities are sustainable and that their levels are optimal. However, resources may be sustainably under-utilised. The implications of these assumptions and the effect of their relaxation are shown in Figure 7.1.

With a zero discount rate, the present values of the benefit streams in Figure 7.1 would be ranked as follows: NPV (a) > NPV (b) > NPV (c) > NPV (d). Thus the most optimal and sustainable use path (a) yields the highest value. If resources are under-utilised (path b) or have been mined to low output levels by past over-utilisation (path d), then the valuation exercise is in danger of underestimating the value of the area. If on the other hand, resources use is assessed at a time when resources are being over-utilised at levels above the maximum sustainable yield (path c), then the exercise will result in an overestimation of
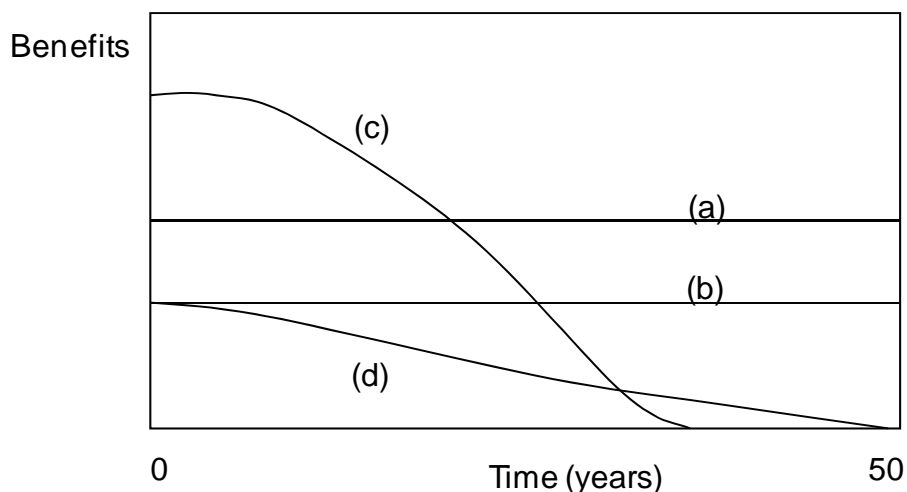
Figure 7.1: Hypothetical, undiscounted benefit stream from a flow of consumptive use of natural resources under base-year conditions of (a) optimal sustainable use, (b) sustained underutilisation, (c) early-stage overexploitation and (d) long-term overexploitation.

the value. To some extent, the effects of over- or under-utilisation may also be reflected in relatively high and low prices and input costs respectively. It is interesting to note that, even if the future path of the net benefits of resource use were known, a high discount rate would tend to favour the over-utilisation of resources. Thus, with a positive discount rate, the present value of path (c) may be higher than the present value of the sustainable path (a), because future benefits in path (a) will be worth very little to the present generation.

Thus, in environmental valuation studies it is imperative that the level of use in relation to optimal sustainable yields is investigated in order to produce a valid interpretation of the results of the valuation methods applied, and a realistic estimate of net present value. The determination of optimal yields requires detailed biological information on the dynamics of resource availability as well as use. A study is underway as part of the Wetland Health and Importance Programme to develop a method for assessing the sustainability of wetland use.

7.4 Estimating contribution to the national economy as income or assets

In some cases, it might be pertinent to express the values in a way that is compatible with national accounting systems, as annual contributions to national income, or in terms of asset values. In addition to wetland valuation for national accounting purposes, this may also include valuation for the justification of conservation action where the impacts are felt on a major scale.

A country's economic performance is measured in terms of its national income and asset base, and the average income per capita is a common (though arguably inadequate) indicator of societal wellbeing. National income is calculated in the National Accounting process, which generates various measures of income such Gross Domestic Product (GDP), Net Domestic Product (NDP) or Net National Product (NNP). The National Accounts quantify the value of capital assets (asset accounts) and the annual value of production (production accounts) at a national scale. As a supplement to the national accounts, many countries have now also developed a number of Natural Resource Accounts (NRA) for various natural assets such as water and minerals (see Lange *et al., 2*003). Thus far, South Africa has developed a preliminary set of water accounts, but this has not yet been extended to include wetlands. Natural resources are not conventionally included as assets in the national accounts, but the NRA supplementary data is very useful in assisting with sustainable development planning. The NRA production accounts measure the use value, in terms of contribution to Gross National Product, of the natural resources each year, and as

such are normally included in the national accounts. The NRA asset accounts measure the value of the natural resource stocks as capital assets.

For some types of decisions as well as for natural resource accounting purposes, it will be relevant to value wetlands in terms of how they contribute to the national or regional economy (i.e. to national income or asset values). This can be done by calculating their value in the same way as these accounts are constructed in the production and asset accounts.

7.4.1   Production accounts

In a National Accounting framework, both direct and indirect contributions to the economy are considered. The direct values generated from production, through direct or indirect use of an ecosystem, are the turnover and net income generated. However, these values are only part of the total macro-economic impact of a wetland. These direct values (not to be confused with "direct use value" in the TEV framework) also generate value indirectly.

For example, through crop production or the provision of tourism services, demand is generated for inputs in the rest of the economy. Thus, in order to provide accommodation services to tourists, hotels and lodges must purchase goods and services used as inputs to production, such as food, textiles, petroleum products, thatch for roofing, telecommunications services, etc. Industries supplying these goods and services must, in turn, employ workers and purchase inputs to produce their goods and services. In addition, when people are employed and earn wages, those wages are used to purchase consumption goods, which must be produced, requiring additional employment and generating more income. This indirect effect is sometimes referred to as the "backward linkage" or "upstream linkage" in the supply chain. Thus, even though tourism enterprises may operate in remote areas, they have an impact throughout the entire economy. Similarly, agriculture and other natural-resource-based activities also have upstream linkages.

The total economy-wide impact of a wetland ecosystem is a sum of the direct plus the indirect impacts. The ratio of the total to direct impact (on sectoral output, incomes, employment or any other variable relevant for policy) is called a "multiplier" because it measures how a change (increase or decrease) in one sector's level of activity will affect the entire economy.

In estimating the value added to the economy from a wetland, direct value added is estimated as the turnover attributable to the wetland minus the expenditure on intermediate goods and services, and the total contribution including knock-on effects is estimated using multipliers calculated in an input-output model or Social Accounting Matrix (SAM) of the regional or national economy. These models are not always constructed in a way that is compatible with the typology of values generated by wetlands. Turpie *et al.* (2006b) modified the national SAM of Botswana to incorporate the tourism value of natural assets. To do this they had to produce coefficients by developing enterprise models of various types of enterprises in the tourism sector, building on models that had been developed for a variety of applications (e.g. Ashley *et al., 1*994; Barnes and De Jager, 1996; Ashley and Barnes, 1996; Barnes, 1996). These kinds of macro-economic models have to be used with caution, however, since they contain rigid assumptions, and may not always be sufficiently accurate for predicting the impacts of changes. For this purpose, more sophisticated Computable General Equilibrium models might be better, though more complicated to construct.

7.4.2   Asset accounts

In the NRA system of the UN *et al.* (2003), natural assets are valued according to the predicted flow of economic rent (resource rent) from the asset base. Only those future rents that are feasible, given economic and policy constraints in the national context are included. NRAs are commonly developed for individual resources, such as fish wildlife and forests, to help with sectoral planning. However, they can also be approached from the point of view of land accounts or ecosystem accounts (Weber, 2006). Turpie *et al.* (2006b) used this approach to value the Okavango Delta, calculating resource rent as the gross output less the costs of production plus a reasonable return to capital. This involved making predictions of the likely future streams of resource rents from each activity over a 30 year period and discounting these streams to obtain the asset value (see section on discounting).

7.5   Describing the contribution to poor households and peoples' livelihoods

Macro-economic models can provide some information on the contribution of wetlands to particular types of households in the regional or national economy. Social Accounting Matrices (SAMs), which are included as part of the 1993 revision of the internationally-applied System of National Accounts (UN *et al., 1*993), are designed to examine the distribution of income in the economy. Although probably better used for large-scale values, such as the value generated by the Okavango Delta, which features prominently in

Botswana's economy (Turpie *et al., 2*006b), the SAM can be used to show the extent to which poor households benefit from the wetland, or are affected by changes in wetland value.

Nevertheless, the above type of distributional analysis does not describe the importance of wetlands in contributing to household livelihoods. It is often of particular interest to understand how wetlands contribute to people's livelihoods in the area immediately around the wetland. This is often particularly relevant when the wetland's contribution to the regional or national economy is small. The latter situation might arise if much of the wetland's value lies in its contribution of resources used for subsistence purposes, which are often overlooked in economic analyses due to the tendency to focus on market-based crops and other forms of livelihoods (Delang, 2006a). Yet, these products are often important in allowing local communities to live off lower incomes than they would do if they had to purchase them in markets (Delang, 2006b), and help to see people through periods of famine (Turpie *et al., 1*999). Moreover, where wetlands support important traditional livelihoods, inappropriate management can lead to increased social and political instability (Ruitenbeek, 1992).

The direct impact of wetlands on rural livelihoods can be estimated in terms of the income (subsistence and cash value) generated by agriculture, natural resource harvesting and through tourism. The latter can include salaries and wages by tourism enterprises, as well as the amount paid to local communities in the form of rentals and royalties (Turpie *et al., 2*006b).

Quantifying the value of wetlands to people's livelihoods involves establishing the extent to which wetland goods and services contribute to overall household income. Turpie *et al.* (1999; 2006b) and Turpie (2000) quantified agricultural outputs and other sources of income as well as the value of natural resource use by households in order to put the use of natural resources into perspective. Although this approach gives some idea of the importance of the wetland, estimating the proportion of household income generated by wetland resources does not necessarily provide a full picture of their importance in terms of risk spreading and safety-net function. Quantifying the risk-spreading function might require more in-depth analyses of income security. Where wetland resources are under communal tenure, they can play a crucial safety-net role in allowing poor people to cope with shocks such as loss of employment (Beck and Nesmith, 2001). This value will be most important in areas with little or no access to social welfare (Nhuan *et al., 2*003), but is nevertheless difficult to quantify. A

dependency metric for wetlands is being developed as part of the Wetland Health and Importance Programme (Turpie *et al.* in prep).

8. WETLAND VALUATION IN PRACTICE

8.1   How is valuation used?

Valuation forms the basis of most resource-economics research and application (Figure 8.1). It has played an important role in lobbying the importance of sustainable use and/or conservation of natural resources. Wetland valuation studies can help to demonstrate the contribution that they make to the local, national or global economy, and thus build local and political support for their conservation and sustainable use.   When put in context, an understanding of the value of wetland services also allows diagnosis of the causes of their degradation and loss. This understanding is critical to identifying the problems, opportunities and constraints that would guide planning, resource allocation and environmental management.   It also allows decision makers to better factor wetlands into development planning and project appraisal through more balanced cost-benefit analysis (CBA). Finally understanding of values, their context, and the goals of planning allows the development of incentive measures and financing mechanisms that help to achieve these goals (Emerton, 1998; Turpie *et al., 2*006a).
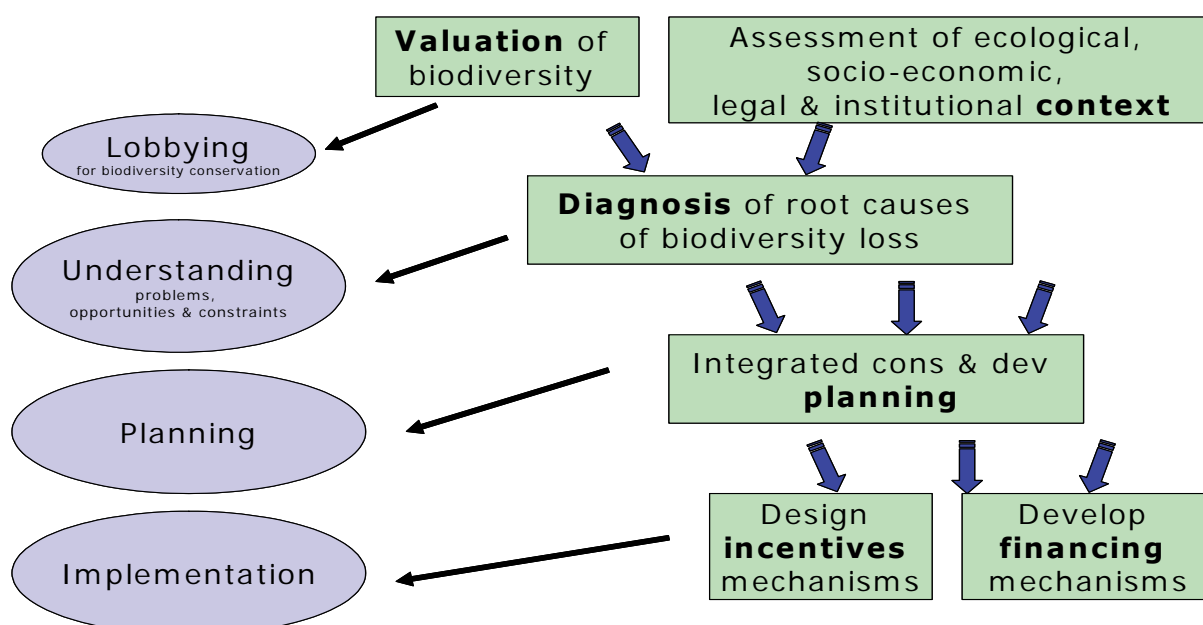
Figure 8.1:   Purposes and applications of the economic valuation and analysis of ecosystems and their biodiversity.

The accurate valuation of wetlands and understanding of their dynamics could be a fundamental prerequisite for the optimisation of planning and management policies and decision-making. The main constraints being that accurate valuation and social assessment methods are:

1. highly specialized, requiring very technical design and analysis;
2. labour-intensive, mostly relying on extensive surveys with large sample sizes; and
3. localised in scope, in that it is very difficult to extrapolate results from one area to the next, something which is could be particularly pertinent in the case of wetlands.

The challenge is therefore to find the right trade-off between accuracy and cost of resource economics studies of wetland ecosystems. Studies need to be focused on the right issues to enable decision-making. Resource economics practitioners will need to devise effective ways of applying their tools which makes them both cost-effective and applicable.

8.2 Valuation for justifying conservation

Much of the early work in wetland valuation was primarily in reaction to the realisation that wetlands were under threat on a large scale, and sought to demonstrate that wetlands had high value. This meant that the main aim of many studies was primarily to articulate their current value in terms of either contribution to the economy or to people's livelihoods. The justification for many of these studies was to demonstrate that the wetlands needed to be conserved or sustainably managed.

In Turpie *et al.*'s (2001) study of the value of Sandvlei, the results suggested that the recreational value exceeded the cost of management of the wetland, and this was used by the City of Cape Town as justification for continued investment in its management.

Terer *et al.* (2004) conducted a study looking at the value of wetland resources to communities around the Tana River National Primate Reserve in Kenya in order to better inform decisions on how to sustainably manage them. It was found that the wetlands provided multiple values and highlighted the important role the local communities needed to play in conservation of the wetlands. Another study on the wetlands of the Kilombero Valley in Tanzania showed that the wetlands are very important in supporting the livelihoods of people living in the valley and drew attention to the potential for resource use conflicts (Kangalawe and Liwenga, 2005).

Turpie *et al.* (1999) highlighted the importance of determining the trajectory of these values in order to ensure that the Zambezi wetlands were sustainably managed. The valuation of the Zambezi basin wetlands (Turpie *et al., 1*999) was carried out in conjunction with an analysis of policy affecting wetlands and was used to inform the development of wetland policy in the region. The Rufiji valuation study (Turpie, 2000) was used in the development of a management plan for the lower basin. In a review of several studies of the value of African wetlands, Schuyt (2005) highlighted their importance to the long term health, safety and welfare of African communities, and called for more sustainable management of these wetlands.

The problem with many such studies, however, is that they fail to elicit the marginal values of these wetlands and their opportunity costs, which would be useful in solving how much to conserve instead of just whether to conserve.

8.3   Valuation for analysing trade-offs

Valuation is increasingly applied in decision-making processes that evaluate the effects (costs and benefits) of alternative development options that affect wetlands. Valuation helps to make more informed decisions and to achieve a solution that provides the highest net benefit. This helps to achieve outcomes that are ecologically sustainable, socially acceptable and economically sound (de Groot *et al., 2*006).

Analysing trade-offs ideally involves developing an understanding of the utility function underlying the wetland value described in TEV studies. In order to measure the impact of change, the utility (value) generated by the wetland can be described as:

$$U = U[X(A), G(A), H(A), Q(A), Z(A), K(A), Y|S]$$

where, U = utility; A is the wetland area; X(A) describes the number of wetland-based recreational trips (e.g. fishing, wildlife observation trips); G(A) is a vector of the resources harvested (e.g. fish caught); H(A) is a vector of non-consumptive quality variables associated with trips (e.g. natural scenery); Q(A) is a vector of off-site recreational use (e.g. watching a television wildlife documentary made on the wetland); Z(A) is a vector of non-recreational on-site and off-site services provided by the wetland (e.g. flood control, groundwater recharge); K(A) is a vector of non-use values associated with wetlands (e.g. existence value); Y represents "all other goods"; and S is a vector of socio-demographic variables (Bergstrom and Stoll, 1993).

8.3.1 *Analysing impacts and land use alternatives*

Wetland valuation may be used to predict the effects of proposed developments on wetlands or alternative management options. It may also be used to examine the impacts of external processes such as climate change on wetland value.

Concern about the lack of consideration of the ecosystem services and other intangible values associated with wetlands in cost-benefit analysis of development projects dates back a long way (Bowers, 1983). Wetlands are threatened by resource use and land-use practices that lead to their degradation and loss. These developments often occur because it is assumed that the damaging activities are worth more than the wetlands themselves. However, several studies have shown that this is not always the case. In Thailand, it was shown that intact mangroves were worth about US$60 000/ha, compared with US$17 000 from shrimp farming, which requires clearing the mangroves (Balmford *et al.,* 2002). In Canada, intact freshwater marshes have been shown to have a higher value than marshes drained for agriculture (US$8800 vs. $3700; Balmford *et al.,* 2002).

Christensen (1982) valued the uses of mangroves for land-use planning in Thailand, Lai (1990) compared the net benefits of converting mangroves to crops in Fiji, and Ruitenbeek (1992, 1994) compared mangrove management options and showed that ecological – economic linkages in a mangrove ecosystem in Indonesia created a case against mangrove clearing in certain areas. Spaninks and van Beukering (1997) valued management alternatives for mangroves in the Philippines. Gilbert and Janssen (1998) assessed the change in goods and services produced by a mangrove ecosystem under different management regimes, but this work was criticised in that it lacked the necessary ecological understanding (Rönnbäck and Primavera, 2000), and could have dire consequences for future decisions about mangroves.

Johnston *et al.* (2002) used a travel cost model to demonstrate how an increase in water quality in the Peconic estuary system, USA, would increase visitor trips to the site, and hence recreational value. Their study illustrated how policy information could be provided through the combination of ecological field measurements (in that case water quality) and economic recreation demand models.

It is important to note that many studies have focussed on comparing total or average values of different land uses. In most cases, it would be pertinent to focus on marginal analysis.

For example, some mangrove might be profitably converted, but the returns to doing so are probably steeply declining.

8.3.2 Conservation and development planning

Successful conservation interventions depend on the availability of reliable information on both their benefits and costs (Kramer and Sharma, 1997; Ferraro, 2002; Balmford *et al.,* 2002; Frazee *et al.,* 2003). Thus broad scale conservation and development planning requires analysis of the benefits and opportunity costs of conservation at a regional scale. However, very few conservation planning studies have considered either economic benefits or opportunity costs (Turner *et al., 2*000; Naidoo and Adamowicz, 2005; Osano *et al.* subm). Furthermore, this type of analysis is potentially complex as the costs and benefits may change depending on the spatial configuration of the conservation or development plan. The incorporation of economic considerations in conservation planning, or conversely, of economic values of ecosystems in development planning, is very new, and there is little precedent at this stage.

In South Africa, Turpie and Clark (2007) have undertaken conservation planning of estuaries which required estimating the total economic value of estuaries under consideration, how these values might be expected to change over time with and without conservation, and the opportunity costs of their conservation. Estimating the economic value of multiple estuaries required taking the unprecedented step of assessing how the physical and locational characteristics of estuaries influenced their value, and using this to estimate values of individual systems. It also required making assumptions, based on scant existing knowledge, about how those values might change under different management scenarios. Finally, opportunity costs had to be estimated in terms of the water that would be held back from other economic uses, as well as the direct management costs. Turpie and Clark (2007) showed that the incorporation of the full suite of costs and benefits greatly alters the choice of configuration of a protected area system when compared to the set of variables that are typically taken into account in conservation planning.

Due to a number of pressures on the Okavango Delta, a very rough analysis of a number of possible future scenarios was undertaken by Turpie *et al.* (2006a), using the results of the valuation study. These included agricultural expansion into the wetland area to meet demands for grazing land, more intensive protection, upstream water abstraction and climate change scenarios. While hydrological models existed to underpin the scenario analysis, expert opinion was used regarding the impacts on biological characteristics. The result

showed that climate change would have a far greater impact on the value of the delta, but that considered; a conservation scenario would have the highest value.

8.3.3   *Water resource planning*

An early study by Barbier *et al.* (1991) on the floodplains of the Hadejia and Jama'are Rivers in Nigeria attempted to estimate the amount of water that needed to be released from upstream developments in order for net economic benefits derived from the floodplain to be maintained.

Emerton (1994) carried out one of the first studies in eastern and southern Africa to estimate the costs that would result from dam construction on the Tana River in terms of loss of wetland value. As the science behind integrated water resource management has escalated in the past decade, so has the development of integrated ecological and economic methods to assess environmental water requirements in order to achieve more socially optimal water allocation decisions. Water resources are scarce throughout southern Africa, and this region has been one of the pioneering areas in wetland valuation for water allocation decision making.

Turpie and Joubert (2001) conducted a conjoint valuation study of the tourism value of rivers in the Kruger National Park, and showed how the model developed in that study could be used to evaluate the impacts of alternative river flow scenarios. Following the development of a framework for determining the ecological water reserve, valuation studies have been conducted of aquatic ecosystems in the Kromme and Seekoei River catchments in South Africa (Turpie, 2006). These studies highlighted the immense information needs required for such a study, especially if it is expected to conduct the studies at a desktop level. Where empirical data collection was possible, Turpie (2006) used a conjoint model to estimate the impacts of different flow scenarios on the recreational value of the Kromme and Seekoei estuaries. The values used were determined by estimates of property values and visitor expenditure based on household surveys. Estimates of the nursery value of these systems were made based on Lamberth and Turpie, 2003, and estimated changes in these values were based on the estimated changes in abundance of the most valuable species by the ichthyologists involved in the ecological studies. These experiences showed that the valuation of river flow scenarios could be achieved through teamwork among specialists of multiple disciplines.

8.4 Contextual issues in valuation

8.4.1 Geographic scale and landscape setting

There are no limits to the spatial extent to which some costs and benefits associated with wetlands could be felt and it is thus important to be explicit about the scale at which benefits and costs are being considered and compared in order to answer the question: "value to whom?". Costs and benefits can be considered at a local, national, regional and global scale. Different values tend to be relevant at different spatial scales (Hein *et al., 2*006; Mitsch and Gosselink, 2000; Soderqvist *et al., 2*000). In general, direct use values are normally considered at the local level, indirect use values at a broader scale, and non-use values at the broadest scale. Local-scale benefits may incur regional-scale costs, and vice versa. 'Local communities' have to be defined on the basis of explicitly stated criteria.

The position of wetlands within a landscape has an influence on value, and therefore on the way in which valuation studies should be approached (Mitsch and Gosseling, 2000; Soderqvist *et al., 2*000). In the case of wetlands, their position within the landscape is closely tied to their type and functioning. For example high altitude wetlands might have a totally different type of interaction with groundwater aquifers than similar wetlands at lower altitude. From reviewing the wetland valuation literature, it is clear that certain types of value are more prevalent in some types of wetlands than others. For example coastal wetlands tend to have high recreational and nursery value, floodplain wetlands are important for flood regulation, and high altitude wetlands for water regulation. It is obviously critical to understand the functional type of a wetland before designing a valuation study. A study conducted in Mpumalanga, South Africa (Palmer *et al., 2*002) highlighted some of the difficulties in this regard. While the area contains thousands of small wetlands of various types, all of these wetlands are interconnected in their hydrological functioning and biodiversity. Since the hydrology of these wetlands was too complex to understand in a reasonable study period, it was not possible to describe their services let alone value them.

8.4.2 Social context and property rights

The value of wetland areas is inextricably linked to the social context in which they occur, and in particular to the property rights assigned to them. Property rights refer to very specific user rights for utilising, conserving or trading particular commodities and assets from an area of land (Bromley, 1991 cited in Adger and Luttrell, 2000). A grasp of the existing property rights and the benefits and constraints these impose upon the people inhabiting the areas around wetlands is extremely important when conducting an economic evaluation.

Property rights can be classified as private property, common property or state property. While the implications of private and state-owned property are straightforward, those of common property resources are less so. Often different wetland features and resources will have their own specific property rights allocated to them. A large number of wetlands are under communal tenure in Africa, including in South Africa. In these wetlands, access to resources is governed by a mixture of state and customary laws. There are various ways that resources may be governed under common property resource use. The one option is a centralised administrative system such as a village committee of elders. Alternatively regulation might be achieved under more diffuse leadership and governors (Adger and Luttrell, 2000). Regulating common property resource use is often continually adjusting with new political and resource use constraints and influences (Shanmugaratnam, 1996).

The extent to which communal property rights are managed depends on various factors such as the level of community cohesion. Under well managed common property systems, individual users tend to have a higher incentive to co-operate with each other than to pursue individualist strategies (Adger and Luttrell, 2000). Under badly-managed systems, individuals have the incentive to pursue their own interests, leading to the loss of overall value (the 'tragedy of the commons' – Hardin, 1968). The success of common property resource management can be assessed by looking at its overall efficiency, how sustainable it is and how equally the different benefits and resources harvested are distributed amongst the community (Adger and Luttrell, 2000).

The level of dependence on natural resources is often high in poverty-stricken communal land areas (Béné, 2003). Poor communities can be defined as those who experience "vulnerability, social marginalization, exclusion from a sustainable livelihood, self-perception of poverty, as well as income poverty" (Beck and Nesmith, 2001). There is a substantial volume of literature supporting the fact that common property resources play a vital role in supporting the poor living in rural communities of developing countries around the world (Beck and Nesmith, 2001). In these communities, natural resources often provide the most significant contribution to people's livelihoods (Beck and Nesmith, 2001). At times the raw materials and basic food commodities provided by an ecosystem may be important to an entire region when no other significant economic activities are taking place (Witt, 2006). It is important to consider that common property resources may play an essential role and be much more pertinent to poor communities than to those from a more affluent background. This will mean that when one is evaluating the value of natural resources from wetlands, some way of quantifying the relative dependency of communities on these resources is needed, as it is likely to vary in different regions and across communities.

When conducting studies investigating resource use by rural communities, the "heterogeneity" of these communities, particularly in their approaches towards generating their own livelihoods, needs to be appreciated (Béné *et al., 2*003). Even within relatively small communities there will be multiple levels of socioeconomic dependency on natural resources and there will be a huge range in different livelihoods and personal incomes. These will be linked with individual's social status and often corresponding access to specific resources (Béné *et al., 2*003). Access to resources can at times be determined by factors such as gender, ethnicity and land ownership. The more destitute households will rely far more heavily upon agriculture and harvesting natural resources, whilst the slightly more affluent families will be able to resort to more reliable sources of income.

1. Many developing economies distort market prices through price fixing or other regulatory activities.
2. Basic data on physical outputs, e.g. fisheries data, is often hard to come by, because it is collected irregularly or unreliably, if at all. Where data is available, it is often stored in a very unprocessed state.
3. Information sources may be biased or conflicting in areas where political and traditional or minority groups do not see eye-to-eye, as is the case in many rural African areas.
4. Inadequate data sources are further exacerbated by difficulty in new data collection. Villages are often remote and access can be difficult and time-consuming or expensive. Due to lack of formal education, and hence a poor understanding of this type of research, rural respondents may often be reluctant to divulge quantitative data to outside researchers. Part of the reason for this may be their reluctance to provide information or opinions when this is the realm of their superiors in the tribal system.
5. Lack of ownership or control of resources may make it difficult for respondents to express willingness to pay in contingent valuation surveys.
6. Gender inequality is an integral part of rural African society. Because of the inferior status of women, their contribution to surveys is very limited. This is a particular problem in the quantification of resource use for resources that are mainly collected by women.
7. Difficulties in gathering survey data may be further exacerbated in some cases by 'survey fatigue', especially in areas where several government and non-government organisations have been active.
8. Cultural differences also hamper the use of hypothetical market surveys. People in rural African communities are not accustomed to handling "what-if" scenarios and are unlikely to provide appropriate answers to such questions. This makes contingent valuation methods or questions about substitution particularly challenging.

9. Cultural differences about the concepts of conservation and development, lead to misunderstanding of research agendas and also hamper the collection of appropriate data.

10. Any surveys in which people are asked to express value in monetary terms are potentially subject to problems when applied in subsistence societies or any communities where money is not the predominant medium of exchange. These problems may be overcome to some extent by research into the barter-exchange value of goods, as discussed above.

11. Lack of markets for communally-owned or state land precludes the use of hedonic pricing methods, which rely on well-functioning property markets.

8.5 Is rapid assessment a viable alternative?

At the same time as methods have become increasingly refined, there has also been pressure to develop rapid, or cheaper, means of assessing the value of ecosystems. Many of the methods described above are extremely data- and labour-intensive. Estimation of direct consumptive and non-consumptive use of natural resources requires surveys of users, and estimation of non-use values also relies on extensive surveys, preferably with sample sizes of over 1000 respondents. The design, execution and analysis of these surveys is a specialised activity and is extremely costly. Hedonic pricing methods to estimate property values are fraught with difficulty in obtaining property value data. The level of difficulty varies between countries, but is fairly high in South Africa. Finally, the estimation of the indirect use values associated with ecosystem functions, which are arguably the most important values of many wetlands, has proved to be extremely difficult for some types of value because of the detailed hydrological and other biophysical data required.

One of the ways that researchers have tried to circumvent these problems is through "benefits transfer" or using values from one system to estimate those of another. The difficulties with this approach are discussed above. Various other rapid assessments have been attempted in southern Africa, with mixed success.

In estimating the value of wetland resource use in tropical wetlands, Turpie *et al.* (1999) tested a rapid approach whereby households were asked to estimate the proportional contribution of different sources using piles of beans. The values assigned to wetland resources in this way were usually similar to those obtained through detailed survey

questionnaires, but there was sufficient variation to cast some doubt on the efficacy of this method. At best it serves to double-check the results obtained from quantitative surveys.

The travel cost method is a data intensive technique and some wetland valuation studies have had to make do estimating tourism value with fewer data. In the valuation of the Okavango Delta, Turpie *et al.* (2006) estimated tourism value in terms of producer surplus, but did not consider consumer surplus due to budgetary constraints and because it was felt that a travel cost analysis would have been hampered by the problem of trips characterised by multiple sites in multiple countries. Thus an inventory of all tourism establishments was made and turnover for the different types of enterprises was estimated on the basis of interviews with tourism operators. A portion of this turnover was then attributed to the delta. Three types of enterprise models were then used: a typical ecotourism lodge, a safari hunting enterprise, and a community-based natural resource management (CBNRM) model to convert turnover estimates into estimates of value added to national income.

Van Zyl and Leiman (2002) tested a short-cut method to estimate the property value of wetlands, by means of estate agent interviews rather than collecting property price data. They concluded that the estate agents were able to help them arrive at estimates that were close to those obtained from hedonic models. However, some authors have shown that hedonic analysis can be made simpler if household data are collected in interviews rather than relying on the deeds office (Boyer and Polasky, 2004; Turpie, 2006b).

With regard to indirect use values, the most expedient estimates appear to be those of replacement costs. The problem with this is that there is no evidence that the service being valued is actually demanded. For example, one might value the flood attenuation capacity of a wetland based on the cost of replacing it with a dam, but in some cases there might be very little of value downstream that is at risk, in which case the rapid method provides an overestimate, or *vice versa*. Although Kotze *et al.* (2008) have developed a scoring system to evaluate wetland ecosystem services based on current understanding, rapid methods to quantify or value the regulating services provided by wetlands have yet to be developed.

In general however, analyses have suggested that rapid assessments should be interpreted with caution. Of particular relevance in this regard, Woodward and Wui (2001) found that, holding all else constant, the values from studies with poor quality econometrics averaged 24-50 times greater than from more rigorous studies.

8.6   What constitutes a credible value for decision-making?

Up to now, policy- and decision-makers have had to welcome and use numerical estimates of the value of ecosystem services almost irrespective of the quality or confidence of the estimation, since these estimates have been fairly hard to obtain. Rough estimates such as those extrapolated from Costanza *et al.*'s (1997) estimates of global average values, have certainly played a role in swaying South African policy makers towards more environmentally conscious thinking.

Broad-scale decision-making can be based on rough estimates, but project level or resource allocation decisions need more reliable estimates. More important is that the confidence of the estimates is known to the decision-makers. The confidence of an estimate can be described in words (e.g. low or high), using ranges of estimates, or by means of statistical confidence intervals. A more sophisticated analysis might involve the use of software such as Excel's @RISK, which calculates a probability distribution for a change in value. Such an analysis would integrate all the uncertainties in the valuation exercise, to give a more realistic idea of the certainty of the result.

8.7   How useful are non-monetary indices such as WET-EcoServices?

A tool has recently been developed for the rapid assessment of the importance of a wetland in terms of its delivery of goods and services by Kotze *et al.* (2008). The tool, called WET-EcoServices, builds on earlier wetland assessment techniques, and takes the form of a series of attributes of the wetland and its catchment that have to be rated on a five-point scale or a binary (yes/no) format. WET-EcoServices flags important ecosystem services that need to be considered in the management of a wetland or in land-use decision processes, but is not designed to provide a single overall measure of value or importance of a wetland, or to quantify (in monetary or other terms) the benefits supplied by a wetland. It only goes as far as to assist in assigning indices to these benefits for comparative purposes (Kotze *et al., 2*008). One of the objectives of this study is thus to evaluate whether WET-EcoServices could provide a useful starting point for a rapid assessment of the social importance or economic value of a wetland.

The services considered in WET-EcoServices are summarised in Table 8.1. Note that all services discussed above apart from groundwater recharge (which is part of the family of flow regulation services) are included.

Table 8.1: Ecosystem services included in WET-EcoServices

Ecosystem services supplied by wetlands	Indirect benefits	Regulating & supporting benefits		Flood attenuation	The spreading out and slowing down of floodwaters in the wetland, thereby reducing the severity of floods downstream
				Streamflow regulation	Influencing the volumes and rates of release of water into a stream or river, sustaining streamflow during low flow periods
			Water quality enhancement benefits	Sediment trapping	The trapping and retention in the wetland of sediment carried by runoff waters
				Phosphate assimilation	Removal by the wetland of phosphates carried by runoff waters, thereby enhancing water quality
				Nitrate assimilation	Removal by the wetland of nitrates carried by runoff waters, thereby enhancing water quality
				Toxicant assimilation	Removal by the wetland of toxicants (e.g. metals, biocides and salts) carried by runoff waters, thereby enhancing water quality
				Erosion control	Controlling of erosion at the wetland site, principally through the protection provided by vegetation.
			Carbon storage		The trapping of carbon by the wetland, principally as soil organic matter
		Biodiversity maintenance			Through the provision of habitat and maintenance of natural process by the wetland, a contribution is made to maintaining biodiversity
	Direct benefits	Provisioning benefits	*Provision of water for human use*		*The provision of water extracted directly from the wetland for domestic, agriculture or other purposes*
			Provision of harvestable resources		*The provision of natural resources from the wetland, including livestock grazing, craft plants, fish, etc.*
			Provision of cultivated foods		*The provision of areas in the wetland favourable for the cultivation of foods*
		Cultural benefits	*Cultural heritage*		*Places of special cultural significance in the wetland, e.g. for baptisms or gathering of culturally significant plants*
			Tourism and recreation		*Sites of value for tourism and recreation in the wetland, often associated with scenic beauty and abundant birdlife*
			Education and research		*Sites of value in the wetland for education or research*

WET-EcoServices divides wetlands into **hydrogeomorphic (HGM) types**, defined on the basis of geomorphic setting (e.g. hillslope or valley bottom; whether drainage is open or closed), water source (surface water dominated or sub-surface water dominated), how water flows through the wetland (diffusely or channelled) and how water exits the wetland (Table

3.3**:** Wetland hydrogeomorphic (HGM) types typically supporting inland wetlands in South Africa.). The assessment can be based purely on a desktop assessment (Level 1 assessment) or be undertaken as a desktop synthesis of available data followed by a rapid field assessment (Level 2 assessment).

Wet Ecoservices considers each service from a supply and demand point of view. The scores reflect:

- **capacity** to provide the service (based on wetland attributes and location); and
- **opportunity** to provide the service (e.g. based on characteristics of surrounding area and population).

First, the potential for the wetland to perform the service is evaluated based on current understanding or expert opinion of wetland characteristics and functioning. Second, the opportunity for the wetland to supply the services is evaluated. For example, the water quality services would not be actualised unless there were anthropogenic inputs in the catchment area or into the wetland. Moreover, the service would not be valuable if there were no beneficiaries downstream. These kinds of factors are all taken into account in the index, which is in line with an economic valuation approach.

There are insufficient data on the value of wetland services in South Africa to perform a statistical analysis of the performance of the index, i.e. whether scores could be correlated with actual value. This index provides a very good reflection of current understanding of wetland functioning and demand for wetland services. There are some considerations, however, that would probably prevent a good relationship between the index value and actual economic value:

The value of some services is likely to be overestimated. It is contentious as to the degree to which wetlands are able to sustain streamflow during low flow periods through the slower release of water. Some hydrological studies have shown this effect to be negligible, and for most wetlands, the delay in flows is likely to be in the order of days rather than months. Groundwater recharge is likely to have a greater impact on downstream low flows. However, since this service has been omitted, it could be combined with the former.

Carbon sequestration is now generally believed to be counterbalanced by methane production, thus scores for this service may be overestimates.

For some services, the size of the wetland relative to the catchment is important. However, its position in the catchment relative to other wetlands is not taken into account.

In compiling the index the assessor is required to describe the activities in the catchment, in order to assess the level of anthropogenic inputs. This type of assessment might be particularly difficult in large catchment areas.

Ascertaining the level of demand for the service is probably more problematic than assessing capacity. For provisioning services, the numbers of nearby households are taken into account. However, even in a level 2 assessment it is difficult to estimate the number of user households around the wetland. The index presents a problem of scale, in that it does not cater for the demand effect of very high numbers of surrounding households, as might be the case for very large wetlands or high population densities.

The above problems could be overcome with refinement and with the use of GIS in conjunction with the tool. However, the overall scoring system presents the most problems.

All supply and demand components of the score are equally weighted. This means that if more factors affect supply than demand, then supply will be more heavily weighted, or vice versa.

All services are scored on the same scale. This is fine if treated in isolation, but it is important to realise that a score of 4 for flood attenuation is not equivalent to a score of 4 for carbon sequestration in terms of value. Thus the score does not necessarily provide an accurate ranking of service values either.

All wetlands are scored on the same scale. This is problematic, in that a score of 4 for wetland A is not likely to be equal to a score of 4 for wetland B in terms of value. This is particularly problematic because wetlands range in size from under a hectare to several thousand hectares.

In general, the principles of the WET-EcoServices are based on sound understanding, but the scaling of the factors contributing to supply and demand would need to be adjusted in the development of a rapid assessment tool that provides a value estimate.

9.   WHAT ARE WETLANDS WORTH?

This section reviews some of the values that are available in the published and grey literature, with particular emphasis on southern and eastern Africa. Further examples of values are found in the list of examples of several wetland studies around the world provided in Appendix 1.

9.1   Direct use values

9.1.1   Natural resource harvesting

Direct use value from the use of natural resources has been estimated for numerous wetlands around the world, including Africa. Some of the pioneering work was done on the Hadejia-Jama'are floodplain in Nigeria (Barbier, 1993). These floodplains are formed where the Hadejia and Jama'are rivers act as tributaries to the Komadugu Yobe River which ultimately drains into Lake Chad. They are used for agriculture, grazing, non-timber forestry produce, firewood and fishing (Barbier, 1993), providing crucial income to the local populace. In this region the wetlands not only benefit the immediate local communities but also serve as grazing lands for semi-nomadic herders and the agricultural surplus provides food for surrounding areas. The use of these wetlands for agriculture, fishing and firewood was valued at approximately US$110-170 per hectare or US$32-49 per 1000 m^3 at around the time of maximum flood inputs (Barbier, 1993).

Much of international work on the value of coastal wetlands and estuaries has concentrated on the value of mangroves (Spaninks and van Beukering, 1997). Bennett and Reynolds (1993) estimated the tourism and fishery values of mangroves in Malaysia; Gammage (1997) valued commercial and community uses of mangroves in El Salvador and Sathirathai (1997) valued a mangrove area in Thailand. In Bintuni Bay, Indonesia, the annual average household income from mangrove wetland sources amounted some US$4500 (Barbier *et al., 1*997). In South Africa, De Wet *et al.* (2005) estimated the use of mangrove harvesting on the Mngazana estuary, Eastern Cape.

Several studies have been carried out on the consumptive use value of wetlands in eastern and southern Africa (Table 9.1). Turpie *et al.* (1999) undertook a comprehensive study of four very large floodplain wetlands within the Zambezi basin: the Barotse wetland (Zambia), eastern Caprivi wetlands (Namibia), Lower Shire wetlands (Malawi) and the Zambezi Delta (Mozambique). In addition to taking advantage of the floodplain productivity for grazing and

agriculture, inhabitants of these wetland areas relied on them for harvesting fish, reeds, sedges, palm leaves, thatching grass, medicinal and food plants, and mangroves and salt at the coast. Similar resources were also harvested from the floodplain wetlands and delta of the Rufiji River system in Tanzania (Turpie, 2000). Some of these resources were used to manufacture a range of products such as sleeping bags, mats, baskets, bed ropes, hats, food covers, fans, ornaments, brooms and grain silos. The wetland resources provided about a quarter of the income to households in the different wetlands, although this varied from site to site. The most pertinent finding of these studies was that the current use value of the wetlands, irrespective of their state of health, was strongly correlated to the number of people that were dependent on them (Turpie *et al., 1*999; Turpie and Barnes, 2003). This was because of the remarkably similar lifestyles of the people in these four countries, which dictated their requirements for natural resources.

Table 9.1: Examples of agricultural and natural resource use values of wetlands from Southern Africa in US$ (excluding recreational uses)

Wetland	Value	Source
Olifants River wetlands, Mpumalanga	Riparian wetlands- $1.4-12/ha/y Seepage wetlands – $209-290/ha/y Pans – $366-378/ha/y Artificial wetlands – $203-215/ha/y	Palmer *et al.,* 2002
Knysna estuary (3594 ha)	$28.3-44.5/ha/y	Napier *et al.,* 2009
Okavango Delta, Botswana (1.3 mha)	$2.33/ha/y	Turpie *et al.,* 2006b
Barotse flood plain, Zambia (550 000 ha)	$15.72/ha/y	Turpie *et al., 1*999
Chobe-Caprivi, Namibia (304 600 ha)	$15.66/ha/y	Turpie *et al., 1*999
Lower Shire, Malawi (243 000 ha)	$81.70/ha/y	Turpie *et al., 1*999
Zambezi Delta, Mozambique (1 789 000 ha)	$6.57/ha/y	Turpie *et al., 1*999
Lake Chilwa wetland, Malawi (240 000 ha)	$85.6/ha/y	Schuyt, 2005
Rufiji floodplain and delta, Tanzania Rivers and lakes – 42 531 ha Floodplain – 179 599 ha Mangroves – 55 154 ha	Rivers and lakes – $42/ha/y Flood plain – $67/ha/y Mangroves – $17/ha/y	Turpie, 2000

Turpie el al. (2006b) estimated the direct use value of resources in the Okavango Delta by the surrounding communities. This wetland was starkly different from those mentioned above in that the level of household income and employment was much higher, a factor attributed to the relative wealth of Botswana and the tourism activities in the area. The majority of households in the area are engaged in farming, including the practice of recession agriculture in "molapos" or pans, which was shown to be 40% more productive per unit area than dryland farming. The reliance on the harvest of natural resources, while significant, is not as high as in other areas, however. This is particularly noticeable in the fishery. Fisheries are well known as "fall-back" resources for poor and disenfranchised households, and in this case the fishery is suspected to be underutilised. In total the wetland is worth some US$3 million per year to households around the Okavango Delta in terms of harvested resources and contribution to agricultural production.

9.1.2   Recreational value

Recreational values can be reflected in property investment to be near to a wetland (property value) or in the expenditure made to visit the wetland (tourism value).

Property values reflect the direct use value of wetlands, usually in terms of recreational or aesthetic value, although they may also reflect wetland contribution to agricultural productivity in the case of farmlands. Numerous studies have estimated the value of urban and coastal wetlands in the developed world using a hedonic property pricing approach (e.g. Doss and Taff, 1996; Mahan *et al., 2*000; Johnston *et al.,* 2002). Studies have also been carried on the property values of urban wetlands and estuaries in South Africa (Van Zyl and Leiman, 2002; Turpie and Joubert, 2004; Turpie, 2006; Table 9.2). All of these studies have found a positive impact of wetlands on property values. Mahan *et al.* (2000) found that both the distance to and size of the nearest wetland affected property prices in North America, with an acre increase in size being valued at $35. It also found that type of wetland was important, with streams being more valuable than lakes. Hedonic studies of the value of wetlands in rural areas of the USA have had mixed results (e.g. Shultz and Taff, 2004; Bin and Polasky, 2004; Boyer and Polask,y 2004). Reynolds and Regalado (2002) found that whether the wetland had a positive or negative impact depended on the wetland type. The impact of wetlands on rural property prices has not been studied in southern Africa.

The tourism value of a wetland is generally estimated using the travel-cost method. Bergstrom *et al.* (1990) were among the first to demonstrate the recreational value of wetlands using the Travel Cost Method, based on extensive survey data, and highlighted

that recreational functions of wetlands should be an important consideration for wetlands policy and management. Numerous such studies have been carried out internationally. However, many attempts at using the Travel Cost Method have been less successful due to the complications of multiple destination visits. The models generated generally have a poor fit even if they are statistically significant. This leads to a high degree of error in the estimation of consumers' surplus. The method is complex and can easily be misunderstood by new practitioners. Turpie *et al.* (2001) provide an uncomplicated example of the use of the Travel Cost Method in estimating tourism value of the Sandvlei estuary.

Tourism value has been found to be negligible for many of the large floodplain wetlands in eastern and southern Africa (Turpie *et al., 1*999; Turpie, 2000). However, a study of the Okavango delta in Botswana showed that the tourism value of this wildlife-rich wetland was the most important direct use value. This study did not have the resources to employ the travel cost method, and instead estimated the total expenditure on tourism, by estimating the turnover at accommodation establishments and in related industries. This meant that the estimated tourism value of about US$185 million did not include the consumers' surplus, and was thus a minimum estimate of the wetland's recreational value.

Estimates of property value in southern Africa range from $47 200 to $80 900/ha, while estimates of tourism value are more variable, ranging from $159-40 440/ha (Table 9.2).

Table 9.2: Examples of recreational and tourism values of wetlands from Southern Africa in US$. Note that the property values are capital values, whereas the tourism values are annual values

Wetland	Property value	Tourism value	Source
Cape Town metropolitan wetlands	$47 200/ha	$220-500/ha/y	Turpie *et al.,* 2001
Sandvlei, Cape Town (155 ha)	$63 200/ha	$525/ha/y	Turpie *et al.,* 2001; van Zyl and Leiman, 2002
Knysna estuary (3594 ha)	$56 619-80 884/ha	$40 442/ha/y	Turpie and Joubert, 2004
Linyati-Chobe, Zambezi Basin	N/a (protected area)	$0.66/ha/y	Seyam *et al.,* 2001
Okavango Delta, Botswana (1.3 million ha)	N/a (communal land)	$159/ha/y	Turpie *et al.,* 2006b

9.2   Indirect use values

A vast amount of literature has accumulated on the valuation of wetland ecosystem services (e.g. Batie and Shabman, 1982; Costanza *et al., 1*989; Barbier, 1993; Barbier *et al., 1*997; Spaninks and Van Beukering, 1997; World Bank, 1998; Emerton, 1998; Turpie *et al., 1*999; Acharya, 2000; Mitsch and Gosselink, 2000).   Indirect use values are more difficult to conceptualise and measure than direct use values.   Indeed, most studies highlight the difficulties in measurement of at least some components of indirect use value because of the considerable amount of biophysical information that is required.   Thus in practice, most empirical studies do not value *all* indirect uses, and where they value, they use cost-based methods, and sometimes the effects on (lost) production.   Nevertheless, the available studies suggest that indirect use values are significant and often similar to or sometimes even larger than the direct use values.

Much of the earliest work on valuing wetland ecosystem services was concentrated on coastal wetlands (Farber and Costanza, 1987, Costanza *et al., 1*989).   Some 25 years ago, Lynne *et al.* (1981) estimated the value of marsh areas in terms of their inputs into marine production processes.   Farber and Costanza (1987) drew attention to the value of coastal wetlands in terms of protection against hurricane damage, and Adger *et al.* (1997) estimated the value of mangroves for coastal protection.   Some of the better known initial work on the valuation of wetland services was on the valuation of mangrove ecosystems as nursery areas for fisheries (Ruitenbeek, 1994).   In South Africa, Lamberth and Turpie (2003) estimated the indirect use value of estuaries as nursery areas to inshore marine commercial and recreational fisheries.   Their estimate was based on the known catches and value of these fisheries, and the level of dependence of each species on estuaries.

Later work on tropical wetlands was also seminal in the progression of this type of research (e.g. Barbier, 1994; Acharya, 2000).   Barbier *et al.* (1997) estimated that mangroves in Bintuni Bay, Indonesia, contributed some 1.9 million Rupees per household per annum in terms of contribution to local agricultural production.   The Hadejia-Jama'are floodplain in Nigeria is important for replenishing a major aquifer in the Chad Formation.   Hydrological studies showed that there was a significant loss of groundwater storage and aquifer recharge from a decrease in the available floodplain area due to development of water resources upstream. This was shown to have significant effects on many of the surrounding villages throughout the region who are dependent upon the aquifer for their domestic water supply and agriculture. Acharya and Barbier (2000) calculated the loss of welfare due to a change in groundwater availability for agriculture, while Acharya and Barbier (1998) valued

the loss of availability of groundwater for domestic consumption. These estimates were made using agricultural production and household production models, the data for which were obtained via household surveys.

Gren *et al.* (1994) attempted to assess the value of services that had been lost due the drainage of wetlands in Europe, by estimating mitigative or avertive expenditure. The study indicated that the annual cost of replacing the wetland's services was between $350 000 and $1 million. Emerton *et al.* (1998) produced an influential study which demonstrated the value of the Nakivubo urban wetland in Uganda, highlighting its important role in treating waste water from Kampala. Turpie *et al.* (2001) also examined the water storage and purification of urban wetlands in Cape Town, South Africa, estimating that they produced annual engineering cost savings in the order of R20 000/ha. These estimates relied on expert understanding of the capacity of these wetlands to remove pollutants.

Five ecosystem services were valued in the case of the Zambezi basin wetlands (Turpie *et al., 1*999): flood attenuation, groundwater recharge, sediment retention, water purification and carbon sequestration. Together, these services were valued at over $182 million per annum. The indirect use values were significant, but were lower than the direct use values of those wetlands. Studies from other parts of the world also demonstrate the significance of indirect use values of wetlands. In New Zealand the indirect use values (storm protection, flood control, habitat, nutrient recycling and waste treatment) of freshwater wetlands in Waikato region were estimated to be $1.2 billion or $39 800/ha (the highest land value after estuaries). Water based ecosystems contributed twice the value of land based systems (Waikato Regional Council, 2006).

In Uganda, the indirect use value of inland water resources, in terms of forest catchment protection, erosion control and water purification, is estimated to be US$300 million per annum (SIWI, 2004).

Four main ecosystem functions were identified as being important in the generation of indirect use value in the Okavango Delta: groundwater recharge, wildlife refuge, carbon sequestration and water purification (Turpie *et al., 2*006b). The Okavango Delta provides a conduit for the recharge of groundwater aquifers which are utilised around the perimeter of the wetland. Some 5.8 mm^3 of groundwater is extracted from the study area, worth an estimated P16 million (Botswana Pula) in terms of market prices. Carbon sequestration was estimated using published rates of sequestration applied to different habitat types, and using published values of carbon, as about P86 million. Wildlife refuge value was estimated by

determining the value of animals that were hunted beyond the delta but whose presence in those areas was attributed to the delta. This service is estimated to contribute P30 million to the hunting industry. Water purification value was estimated by calculating the input of pollutants and estimating what the artificial treatment cost of this quantity of effluent would be. Relatively little wastewater finds its way into the wetland, however, and the service is valued at about P2.2 million.

Values obtained for ecosystem services in southern Africa are summarised in Table 9.3. These range from less than $100/ha/y for some of the larger systems in other countries, to over $2000/ha/y for coastal and urban wetlands in South Africa.

Many of the estimates of indirect use value, including those in Table 9.3 remain controversial. The estimation of indirect use values requires in-depth understanding of the ecosystem under review, and inadequate ecological knowledge is often a constraint for their estimation. In the absence of the required ecological knowledge, assumptions need to be made in order to estimate values.

Table 9.3: Examples of indirect use values of wetlands from Southern Africa in US$

Wetland	Type of service	Value	Source
Cape Town metropolitan wetlands	Water storage and purification function	$2 100-2 325/ha/y	Turpie *et al.,* 2001
Knysna estuary (3594 ha)	Fish nursery area	$5423/ha/y	Turpie and Clark, 2007
Okavango Delta, Botswana (1.3 million ha)	Groundwater recharge	$2.27/ha/y	
	Carbon sequestration	$12.25/ha/y	
	Wildlife refuge	$10.95/ha/y	Turpie *et al.,* 2006b
	Water purification	$0.32/ha/y	
	Education and scientific value	$2.56/ha/y	
Barotse flood plain, Zambia (550 000 ha)	Groundwater recharge, Carbon sequestration + Water purification	$79.82/ha/y	Turpie *et al.,* 1999
Chobe-Caprivi, Namibia (304 600 ha)	Groundwater recharge, Carbon sequestration + Water purification	$72.2/ha/y	Turpie *et al.,* 1999
Lower Shire, Malawi (243 000 ha)	Groundwater recharge, Carbon sequestration + Water purification	$150.6/ha/y	Turpie *et al.,* 1999
Zambezi Delta, Mozambique (1 789 000 ha)	Groundwater recharge, Carbon sequestration + Water purification	$44.66/ha/y	Turpie *et al.,* 1999

9.3   Non-use value

Much less work has been carried out on the non-use value of wetlands than on other types of value. None of the mangrove studies reviewed by Spaninks and van Beukering (1997) included non-use value estimates.

Using contingent valuation, Hammitt *et al.* (2001) produced an extremely high estimate of the non-use value of a coastal wetland in Taiwan as between US$200 million and US$1.2 billion. In comparison, Turpie *et al.* (1999) obtained a value of only $4 million for the much larger Barotse wetland in Zambia (Table 9.4).

Turpie and Savy (2005) estimated the existence value of South African estuaries (R93 million per annum; Table 9.4), and of the Knysna estuary in particular (R9.7 million per annum). Turpie and Clark (2007) found that the existence value of individual estuaries was closely related to their scenic beauty, and not size.

Table 9.4: Examples of non-use values of wetlands from Southern Africa in US$

Wetland	Value	Source
South African estuaries (70 000 ha)	$12.86 m	Turpie and Savy, 2005; Turpie and Clark, 2007
Barotse flood plain, Zambia (550 000 ha)	$4.2 m	Turpie *et al., 1*999

9.4   Total economic value

In 1989, Costanza *et al.* concluded that "no reasonable amount of effort will produce very precise estimates of wetland values". Indeed, all of the types of estimation described above can only be described as rough, at best. Even measures that are achieved through precise models must be considered as having a large margin of error. All of these estimates model or analyse human behaviour, which is variable and unpredictable in many dimensions.

In the effort to value wetlands, the vast majority of studies have not arrived at a total economic value, and can only be considered partial valuations. Values that are omitted from studies are most often ecosystem services for which biophysical understanding is lacking. Rönnbäck and Primavera (2000) highlighted the potential pitfalls of partial valuation in leading to distorted decision-making. This highlights the need for ecological research to support economic valuation studies.

Of even greater concern is the fact that very few studies take valuation a step beyond total economic value to estimate the marginal values involved. This is crucial for estimating the impacts of changes in management or land-use.

9.5   Comparisons between wetland values

The previous sections, as well as the studies listed in Appendix 1, highlight the variability in values between different wetlands as well as the numerous ways in which valuation studies have been approached and the values expressed. In an attempt to compare the values of different wetlands, several authors have conducted reviews and meta-analyses of the value of wetlands, based on the large amount of literature on this topic. Heimlich *et al.* (1998) compared the results of 33 studies conducted over 26 years, and found values ranged from US$0.06 to $22 050 per acre. Batie and Wilson (1978) considered only a single service – the contribution of wetlands to oyster production – and still found values to differ by two orders of magnitude between sites. In comparing the results of 39 wetland valuation studies, Woodward and Wui (2001) found no real trends that could help to predict the value of a wetland, and concluded that there is still a need for site-specific valuation.

10. APPLYING WETLAND VALUATION IN SOUTH AFRICA

The above review highlights some important lessons in applying wetland valuation in South Africa. South Africa has a multitude of wetland types, social contexts and they lie in a variety of geographic and landscape contexts. The problems facing South African wetlands are a mixture of those found in the developed and the developing nations. The decision making contexts, particularly regarding land use, conservation and development planning, and water allocation, are common problems in most of the countries where valuation has taken place. Being a developing country, data availability is often a constraint, and the lack of biophysical data on wetland functioning is probably one of the biggest obstacles to wetland valuation in South Africa.

Thus there is no specific valuation context or situation that is peculiar to South Africa that has not been encountered in wetland valuation studies elsewhere. Thus, in general, wetland valuation should continue to follow best practice for ecosystem valuation. Ideally, this should continue until valuation studies can provide numerous examples of different types and sizes of wetlands in different geographic and social settings. Extrapolation of high-confidence

values would require considerably more comprehensive valuation studies than exist at present.

Nevertheless, there is increasing pressure to develop rapid, cheaper methods in South Africa, particularly with the current emphasis on the determination of environmental flows under the South African National Water Act No. 36 of 1998, but also due to the pressures of development. Up until now, international experience has shown that the use of rapid methods is potentially fraught with inaccuracy, especially regarding the use of benefits transfer. However, there have been some promising studies which suggest that other rapid valuation techniques may be feasible, though these still require some level of data collection or surveys. If a desktop-level rapid valuation method is to be developed for South African wetlands at this stage, it will only be possible at a level that generates low confidence estimates, providing rough ranges of value suitable for coarse-level decision-making.

11. REFERENCES

Abila, R.O.; Othina, A.Ministry of Water and Irrigation, Nairobi (Kenya); (2006), What is the socio-economic value of the wetlands fisheries? The case of Yala Wetland in Kenya , Published in: Odada, Eric & Olago, Daniel O. (Ed.) Proceedings of the 11th World Lakes Conference: vol. 2. p. 178-185.

Abril, A. and Bucher, E.H. 2001. Overgrazing and soil carbon dynamics in the Western Chaco of Argentina. Applied Soil Ecology 16: 243-249.

Acharya G and Barbier EB. 1998. Valuing environmental functions using domestic water demand analysis. Paper presented at the World Congress of Environmental and Resource Economists, June 25-27, 1998 Venice, Italy.

Acharya G and Barbier EB. 2000. Valuing groundwater recharge though agricultural production in the Hadejia-Nguru wetlands in Northern Nigeria. Agricultural Economics 22: 247-259.

Acharya G. 2000. Approaches to valuing the hidden hydrological services of wetland ecosystems. Ecological Economics 35: 63-74.

Adamus, P.R. and Stockwell, L.T. 1983. A Method for Wetland Functional Assessment. Vol. I. Critical Review and Evaluation Concepts. Report No. FHWA-IP-82-23. Federal Highway Administration, Washington, D.C.

Adaya AL, Bdliya H, Bitrus H, Danjaji M, Eaton D, Gambo MB, Goggobe M, Makinta A, Okali D, Omoluabi AD, Polet G, Salisu M, Sanusi SS, Sarch MT and Shuaibu M. 1997. Local-level assessment of the economic importance of wild resources in the Hadejia-Nguru Wetlands, Nigeria. Hidden Harvest Project, Research Series Vol. 3, No. 3.

Adger WN and Luttrell C. 2000. Special issue: The values of wetlands: Landscape and institutional perspectives. Property rights and the utilisation of wetlands. Ecological Economics 35: 75-89.

Adger NW, Kelly M and Hoang Tri N. 1997. Economic value of mangrove restoration as local resource and storm protection in Vietnam. In: Forestry in a Changing Political Environment. Fifteenth Commonwealth Forestry Conference, Victoria Falls, Zimbabwe.

Allan, D.G., Seaman, M.T. & Kaletja, B. 1995. The endorheic pans of South Africa. Pp 75-102 in: Cowan, G.I. (ed.) Wetlands of South Africa. Department of Environmental Affairs and Tourism, Pretoria.

ANON 1993. Wetlands as a Nonpoint Source Pollution Control Measure in Rouge River National Wet Weather Demonstration Program,1993, pp. 5- 9 http://www.rougeriver.com/pdfs/wetlands/tm12.pdf.

Arrow K, Solow R, Portney PR, Learner EE, Radnar R and Schuman H. 1992. Report of the National Oceanic and Atmospheric Administration (NOAA) panel on Contingent Valuation. Washington D.C. 64pp.

Arrow K, Solow R, Portney PR, Learner EE, Radner R and Schuman H. 1993. Report of the National Oceanic and Atmospheric Administration (NOAA) panel on Contingent Valuation. National Oceanic and Atmospheric Administration, Washington D.C. 64pp.

Asheim G. 1997. Adjusting green NNP to measure sustainability. Journal of Economics 99: 335-70.

Ashley C and Barnes J. 1996. Wildlife Use for Economic Gain: The potential for Wildlife to Contribute to Development in Namibia. In: Environmental Sustainability: Practical Global Implications, Smith FDM (ed.). St. Lucie Press. Delray Beach, Florida. USA. pp. 163-191.

Ashley C, Barnes J and Healy T. 1994. Profits, equity, growth and sustainability: The potential role of wildlife enterprises in Caprivi and other communal areas of Namibia. Research Discussion Paper No. 2. Directorate of Environmental Affairs. Ministry of Environment and Tourism. Windhoek, Namibia. 26pp.

Aylward BA and Barbier EB. 1992. Valuing environmental functions in developing countries. Biodiversity and Conservation 1: 34-50.

Baland JM and Platteau JP. 1996. Halting degradation of natural resources: is there a role for rural communities? Clarendon, Oxford. In: Adger, W.N. and Luttrell, C. 2000. Special issue: The values of wetlands: landscape and institutional perspectives. Property rights and the utilisation of wetlands. Ecological Economics 35: 75-89.

Balmford A, Bruner A, Cooper P, Costanza R, Farber S, Green RE, Jenkins M, Jefferiss P, Jessamy V, Madden J, Munro K, Myers N, Naeem S, Paavola J, Rayment M, Rosendo S, Roughgarden J, Trumper K and Turner RK.   2002. Economic reasons for conserving wild nature. Science 297: 950-53.

Balmford A, Gaston JK, Blyth S, James A and Kapos V. 2003. Global variation in terrestrial conservation costs, conservation benefits, and unmet conservation needs. PNAS 100:1046-1050.

Barbier EB.   1993. Sustainable use of wetlands – valuing tropical wetland benefits: Economic methodologies and applications. The Geographical Journal 159: 22-32.

Barbier EB. 1994. Valuing environmental functions: Tropical wetlands. Land Economics 70: 155-173.

Barbier E, Acreman MC and Knowler D. 1997. Economic Valuation of Wetlands: A Guide for Policy Makers and Planners. Ramsar Convention Bureau, Switzerland. 127pp.

Barbier EB, Adams WM and Kimmage K.   1991. Economic valuation of wetland benefits: The Hadejia-Jama'are floodplain, Nigeria. International Institute for Environment and Development, UCL London Environmental Economics Centre, Paper DP 91-02.

Barnes JI.   1988. Wildlife conservation and utilisation as complements to agriculture in southern African development. DEA Research Discussion paper 27. 20pp.

Barnes JI.   1995. Economic analysis of community-based wildlife utilisation initiatives in Botswana. Development Southern Africa 12: 783-803.

Barnes JI.   1996. Economic characteristics of the demand for wildlife viewing tourism in Botswana. Development Southern Africa 13: 377-397.

Barnes JI.   1998. Wildlife economics: a study of direct use values in Botswana's wildlife sector. PhD Thesis, University College, University of London, London, UK. 370pp.

Barnes JI. 2006. Economics analysis of human – elephant conflict in the Okavango Delta Ramsar site, Botswana. Unpublished report.

Barnes JI, Cannon J and Morrison K. 2001. Economic Returns to Selected Land Uses in Ngamiland, Botswana. Unpublished Report, Conservation International, Washington, D.C., USA. 166pp.

Barnes JI and De Jager JLV. 1996. Economic and financial incentives for wildlife use on private land in Namibia and the implications for policy. South African Journal of Wildlife Research. 26: 37-46.

Barry D and Oelschlaeger M. 1996. A science for survival: Values and conservation biology. Conservation Biology 10: 905-11.

Barton D. 2002. The transferability of benefits transfer. Ecological Economics 42: 147-164.

Bates LJ and Santerre RE. 2001. The public demand for open space: The case of Connecticut communities. Journal of Urban Economics 50: 97-111.

Batie SS and Shabman LA. 1982. Estimating the economic value of wetlands: Principles, methods and limitations. Coastal Zone Management Journal 10: 255-278.

Batie SS and Wilson JR. 1978. Economic values attributable to Virginia's coastal wetlands as inputs in oyster production. Southern Journal of Agricultural Economics 10: 111-118.

Batty, L.C., Atkin, L. and Manning, D.A.C. 2005. Assessment of the ecological potential of mine-water treatment wetlands using a baseline survey of macroinvertebrate communities. Environmental Pollution 138: 412-419.

Beck T and Nesmith C. 2001. Building on poor people's capacities: The case of common property resources in India and West Africa. World Development 29:19-133.

Behling, H. 2002. Carbon storage increases by major forest ecosystems in tropical South America since the Last Glacial Maximum and the early Holocene. Global and Planetary Change 33: 107-116.

Béné C. 2003. When fishery rhymes with poverty: A first step beyond the old paradigm on poverty in small-scale fisheries. World Development 31: 949-975.

Béné C, Allison EH, and Horemans B. 2006. Vulnerability and poverty among small-scale fisher folk. 1-12. In: Witt, R. Food security and poverty alleviation through improved valuation and governance of river fisheries in Africa. Research proposal. University of Hanover, Germany.

Béné C, Neiland A, Jolley T, Ovie S, Sule O, Ladu B, Mindjimba K, Belal E, Tiotsop F, Baba M, Dara L and Zakara A. 2003. Inland fisheries, poverty and rural livelihoods in the Lake Chad Basin. Journal of Asian and African Studies 38: 17-51.

Benessaiah N. 1998. Merja Zerga. In: Mediterranean wetlands, socio-economic aspects. Ramsar Convention Bureau, Gland, Switzerland. pp. 65-70.

Bennett, EL and Reynolds CJ 1993. The value of a mangrove area in Sarawak. Biodiversity and Conservation 2:359-375.

Bergkamp, G. and Orlando, B. 1999. Wetlands and climate change: exploring collaboration between the Convention on Wetlands (Ramsar 1971) and the UN framework Convention on Climate Change, October 1999.

Bergstrom JC and Stoll JR. 1993. Value estimator models for wetlands-based recreational use values. Land Economics 69: 132-137.

Bergstrom JC, Stoll JR, Titre JP and Wright VL. 1990. Economic value of wetlands-based recreation. Ecological Economics 2: 129-147.

Beringer, J., Hutley, L.B. , Tapper, N.J. and Cernusak, L.A. 2007. Savanna fires and their impact on net ecosystem productivity in North Australia. Global Change Biology 13: 990-1004.

Bin O and Polasky S. 2004. Effects of flood hazards on property values: evidence before and after hurricane Floyd. Land Economics 80: 490-500.

Bingham G, Brody M, Bromley D, Clark E, Cooper W, Costanza R, Hale T, Hayden G, Kellert S, Nargaard R, Norton B, Payne J, Russell C and Suter G. 1995. Issues in ecosystem valuation: Improving information for decision making. Ecological Economics 14: 73-90.

Birol E, Karousakis K and Koundouri P. 2006. Using a choice experiment to account for preference heterogeneity in wetland attributes: The case of *Cheimaditida* wetland in Greece. Ecological Economics 60: 145-156.

Bishop RC and Heberlein TA. 1979. Measuring values of extra-market goods: Are indirect measures biased? American Journal of Agricultural Economics 61: 926-930.

Bockstael NE. 1995. Travel cost models. Pp. 655-671. In Bromley DW (ed). Handbook of environmental economics. Oxford, Blackwell Publishers.

Bolt K, Ruta G and Sarraf M. 2005. Estimating the cost of environmental degradation. The World Bank. 265pp.

Bolund P and Hunhammar S. 1999. Ecosystem services in urban areas. Ecological Economics 29: 293-301.

Boto K.G. and Patrick W.H. 1979. Role of wetlands in removal of suspended sediments. In: Greeson P.E., Clark J.R. and Clark J.L. (eds.) Wetland functions and values: the state of our understanding. American Water Resources Association, Minneapolis, Minnesota.

Bowers J. 1983. Economics and conservation: The case of land drainage. In: Conservation in perspective, Warren A and Goldsmith FB. John Wiley and Sons Ltd. pp. 375-389.

Boyer, T., and S. Polasky. 2004. Valuing urban wetlands: a review of non-market. valuation studies. Wetlands 24, 4: 744-755.

Breen CM and Begg GW. 1989. Conservation status of Southern African wetlands. In: Biotic diversity in Southern Africa: Concepts and conservation. Huntley BR (ed.) Conservation Biology 14: 1490-1499.

Breffle WS, Morey ER and Lodder TS. 1998. Using contingent valuation to estimate a neighbourhood's willingness to pay to preserve undeveloped urban land. Urban Studies 35: 715-727.

Brinson MM. 1993. A hydrogeomorphic classification for wetlands. Wetlands Research Program Technical Report WRP-DE-4. 103pp.

Brinson, M.M. 2000. Changes in the functioning of wetlands along environmental gradients. Wetlands 13: 65-74.

Briscoe J, Furtado de Castro P, Griffin C, North J and Olsen O. 1990. Toward equitable and sustainable rural water supplies: A contingent valuation study in Brazil. The World Bank Economic Revue 4: 115-134.

Bromley DW.   1991. Environment and economy: Property rights and public policy. Blackwell, Oxford.   In: Adger WN and Luttrell C.   2000. Special issue: The values of wetlands: Landscape and institutional perspectives.   Property rights and the utilization of wetlands. Ecological Economics 35: 75-89.

Burgess J, Clark J and Harrison CM.   2000. Special issue, the Values of wetlands: Landscape and institutional perspectives. Knowledge in action: An actor network analysis of a wetland agri-environment scheme. Ecological Economics 35: 119-132.

Calvo C and Dercon S. 2005. Measuring individual vulnerability. Discussion Papers Series, Department of Economics, Oxford University 229: 1-30.

Carlsson F, Frykblom P and Liljenstolpe C. 2003. Valuing wetland attributes: An application of choice experiments. Ecological Economics 47: 95-103.

Cao, M., Marshall, S. and Gregson, K. 1996. Global carbon exchange and methane emissions from natural wetlands: Application of a process-based model, *J. Geophys. Res.*, 101: 14,399-14,414.

Carson RT.   1991. Constructed Markets. Pp. 121-162 In: Measuring the Demand for Environmental Quality.   Braden JB and Kolstad CD (eds.). Elsevier Science Publishers, North Holland.

Carson RT, Flores NE, Martin KM and Wright JL. 1996. Contingent valuation and revealed preference methodologies: Comparing the estimates for quasi-public goods.   Land Economics 72: 80-96.

Carson RT, Flores NE and Meade NF. 2001. Contingent valuation: Controversies and evidence. Environmental and Resource Economics 19: 173- 210.

Carter, V., Bedinger, M.S., Novitzki, R.P. and Wilen, W.O. 1979. Water resources and wetlands. Pp 344-376 in: Greeson, P.e., Clark, J.R. and Clark J.E. (eds). Wetland functions and values: the state of our understanding. Proc Nat Symp on Wetlands, Lake Buena Vista, Fl. American Water Resources Association. Minneapolis, MN.

Clawson M. and Knetsch J.L. 1966. Economics of outdoor recreation. Baltimore: Johns Hopkins University Press for Resources for the Future.

Cooper J and Loomis J. 1992. Sensitivity of willingness-to-pay estimates to bid design in dichotomous choice contingent valuation models. Land Economics 68: 211-224.

Costanza R, d'Arge R, De Groot R, Farber S, Grasso M, Hannon B, Limburg K, Naeem S, O'Neill RV, Paruelo J, Raskin RG, Sutton P and van den Belt M. 1997. The value of the world's ecosystem services and natural capital. Nature 387:253-259.

Costanza R, Farber SC and Maxwell J. 1989. Valuation and management of wetland ecosystems. Ecological Economics 1: 335-361.

Cowan GI. 1995. Wetland regions of South Africa. In: Wetlands of South Africa. Cowan, G.I. (ed.) Department of Environmental Affairs and Tourism, Pretoria, South Africa. 292pp.

Cowardin LMV, Carter V, Golet FC and La Roe ET. 1979. Classification of wetlands and deepwater habitats of the United States. US Fish and Wildlife Service FWS/OBS-79/31, Washington D.C. 79pp.

Craft, C.B. and Casey, W.P. 2000. Sediment and nutrient accumulation in floodplain and depressional freshwater wetlands of Georgia, USA. Wetlands 20(2):323-332.

Daily GC. (ed.) 1997a. Nature's Services: Societal Dependence on Natural Systems. Island Press, Washington D.C. 392pp.

Daily GC. 1997b. Introduction: What are Ecosystem Services? In: Nature's Services: Societal Dependence on Natural Ecosystems, G.C. Daily (ed.), Island Press, Washington D.C. pp. 1-10.

Daily GC, Söderqvist T, Aniyar S, Arrow K, Dasgupta P, Ehrlich PR, Folke C, Jansson AM, Jansson BO, Kautsky N, Levin S, Lubchenco J, Mäler KG, Simpson D, Starrett D, Tilman D and Walker B. 2000. The value of nature and the nature of value. Science 289: 395-396.

Danielson LE and Leitch JA. 1986. Private vs. public economics of prairie wetland allocation. Journal of Environmental Economics and Management 13: 81-92.

de Groot RS. 1992. Functions of Nature: Evaluation of Nature in Environmental Planning, Management and Decision-Making. Wolters Noordhoff BV, Groningen, the Netherlands. 345pp.

de Groot RS, Wilson M and Boumans R. 2002. A typology for the description, classification and valuation of ecosystem functions, goods and services. In: The dynamics and value of ecosystem services: integrating economic and ecological perspectives. Ecological Economics 41: 367-567. pp. 393-408.

de Groot RS, Stuip M, Finlayson M and Davidson N. 2006. Valuing Wetlands: Guidance for Valuing the Benefits Derived from Wetland Ecosystem Services. Ramsar Technical Report no. 3. CBD technical series no. 27. Gland, Switzerland. 54pp.

de Groot RS, van der Perk JP, Chiesura A and van Vliet AJH. 2003. Importance and threat as determining factors for criticality of natural capital. Ecological Economics 44: 187-204.

Delang CO. 2006a. Economic valuation of non-marketed wild edible plants in Thailand. Environmental Conservation 32: 285-287.

Delang CO. 2006b. Analysis: Not just minor forest products: The economic rationale for the consumption of wild food plants by subsistence farmers. Ecological Economics 59: 64 -73.

Desvouges, WH, Johnson, F, Dunford, R, Hudson, S, Wilson, K, and Boyle, K (1993). Measuring resource damages with contingent valuation. In J. A. Hausman (Ed.), Contingent valuation: A critical assessment (pp. 91-159). Amsterdam: North Holland.

De Wet J, Turpie JK and Schroen J. 2005. The value of the Mngazana estuary mangroves to local communities in the eastern cape, South Africa. In Turpie, J.K. & Hosking, S.H. (eds) Proceedings of a national workshop on resource economics as a tool for the management and conservation of estuaries in South Africa. Anchor Environmental Consultants.

Diamond PA and Hausman JA. 1994. Contingent valuation: Is some number better than no number? Journal of Economic Perspectives 8: 45-64.

Dixon J and Pagiola S. 1998. Economic analysis and environmental assessment. Environmental assessment sourcebook update, April 1998, Number 23. Environment Dept., the World Bank. 14pp.

DossCR and Taff SJ. 1996. The influence of wetland type and wetland proximity on residential property values. Journal of Agricultural and Resource Economics 21.

Eaton D and Sarch M-T. 1997. The Economic Importance of Wild Resources in the Hadejia-Nguru Wetlands, Nigeria. Amsterdam, Vrije Universiteit, Amsterdam: 1-41.

Elder JF and Goddard GL. 1996. Sediment and Nutrient Trapping Efficiency of a Constructed Wetland near Delavan Lake, Wisconsin, 1993-1995. US Geological Survey Fact Sheet FS-232-96.

Ellis GM and Fisher AC. 1987. Valuing the environment as input. Journal of Environmental Management 25: 149-56.

Emerton L. 1994. An Economic Valuation of the Costs and Benefits in the Lower Tana Catchment Resulting from Dam Construction. Nippon Koei and Acropolis Ltd, Nairobi.

Emerton L. 1998. Economic Tools for Valuing Wetlands in Eastern Africa, Biodiversity Economics Manual: IUCN Eastern Africa Regional Office, Nairobi.

Emerton L and Bos E. 2004. Value – counting ecosystems as an economic part of water infrastructure. IUCN, Gland, Switzerland and Cambridge, UK. 88pp.

Ewart-Smith JL, Ollis DJ, Day JA and Malan HL. 2006. National Wetland Inventory: Development of a Wetland Classification System for South Africa. Report no. KV 174/06, Water Research Commission, Pretoria. 103pp.

Ewel C. 1997. Water quality improvement by wetlands. In: Daily G. (Ed.) *Nature's Services: Societal Dependence on Natural Ecosystems*, Island Press: Washington DC.

Fankhauser, S. and Tol, R.S.J. 1997. The social costs of climate change: the IPCC second assessment report and beyond. *Mitigation and Adaptation Strategies for Global Climate Change* **1**: 385-/403.

Farber S and Costanza R. 1987. The economic value of wetland systems. Journal of Environmental Management 24: 41-51.

Farber SC, Constanza R and Wilson MA. 2002. Economic and ecological concepts for valuing ecosystem services. Ecological Economics 41: 375-92.

Faulkner, S.P. and Richardson, C.J. 1989. Physical and Chemical Characteristics of Freshwater Wetland Soils. Pp 41-72 In Hammer, D.A. Constructed wetlands for wastewater treatment: municipal, industrial, and agricultural. Chelsea, MI: Lewis Publishers.

Ferraro PJ. 2002. The local costs of establishing protected areas in low-income nations: Ranomafana national park, Madagascar. Ecological Economics 43:261-275.

Forbes, G., Dickson, K.R., Golden, T.D., Hudak P. and Doyle, R.D. 2004. Dissolved phosphorus retention of light-weight expanded shale and masonry sand used in subsurface flow treatment wetlands. Environmental Science Technolology 38: 892-898b

Frazee SR, Cowling RM, Pressey L, Turpie JK and Lindenberg N. 2003. Estimating the costs of conserving a biodiversity hotspot: a case-study of the Cape Floristic Region, South Africa. Biological Conservation 112:275-290.

Gammage S. 1997. Estimating the returns to mangrove conversion: Sustainable management or short term gain? IIED Environmental Economics Discussion Paper, DP97-02.

Gersberg, R.M., Brenner, R., Lyon, S.R. and Elkins, B.V. 1987. Survival of bacteria and viruses inmunicipal wastewater applied to artificial wetlands . Pp 237-245 In: Reddy, K.R. & Smith, W.H. (eds). Aquatic plants for water treatment and resource recovery. Magnolia Publishing, Orlando Florida.

Georgiou S, Whittington D, Pearce D and Moran D. 1997. Economic Values and the Environment in the Developing World.

Gilbert AJ and Janssen R. 1998. Use of environmental functions to communicate the values of a mangrove ecosystem under different management regimes. Ecological Economics 25: 323-346.

Green PE and Rao VR. 1971. Conjoint measurement for quantifying judgement data. Journal of Market Research 8: 355-363.

Gren IM, Folke C, Turner K and Bateman I. 1994. Primary and secondary values of wetland ecosystems. Environmental and Resource Economics 4: 55-74.

Hammit JK, Liu JT and Liu JL. 2001. Contingent valuation of a Taiwanese wetland. Environment and Development Economics 6: 259-268.

Hanley N, Wright RE and Alvarez-Farizo B. 2006. Estimating the economic value of improvements in river ecology using choice experiments: An application to the water framework directive. Journal of Environmental Management 78: 183-193.

Hardin G. 1968. The tragedy of the commons. Science 162: 1243-1248.

Hartwick J. 1994. National wealth and net national product. Scandinavian Journal of Economics 99: 253-56.

Heimlich R, Wiebe K, Claassen R, Gadsby D and House R. 1998. Wetlands and Agriculture: Private Interests and Public Benefits. AER-765. U.S. Dept. Agr., Econ. Res. Serv.

Hein L, van Koppen K, De Groot RS and van Ierland EC. 2006. Spatial scales, stakeholders and the valuation of ecosystem services. Ecological Economics 57: 209-228.

Helliwell DR. 1969. Valuation of wildlife resources. Regional Studies 3: 41-49.

Helm D. 1996. Economic Policy Towards the Environment. Blackwell, Oxford.

Hernandez-Cornejo R and Ruiz-Luna A (eds.). 2000. Development of shrimp farming in the coastal zone of southern Sinaloa (Mexico): Operating characteristics, environmental issues and perspectives. Ocean and Coastal Management 43: 597-607.

Hoekstra AY, Savenije HHG and Chapagain AK. 2001. An integrated approach towards assessing the value of water: a case study on the Zambezi basin. Integrated Assessment: 2: 199-208.

IIED 1997. Valuing the hidden harvest: Methodological approaches for local-level economic analysis of wild resources. Sustainable Agriculture Research Series 3 (4). Sustainable Agriculture Programme, IIED, London.

Jacobs M. 1997. Environmental Valuation, Deliberative Democracy and Public Decision-Making. In: Valuing Nature: Economics, Ethics and Environment. Foster J (ed.). Rutledge, London. pp. 211-231.

Jacobsen T, MacDonald A and Enggrob H. 2005. ODMP Hydrology and water resources: integrated hydrological model. DWA and DANIDA. 64pp.

James RF. 1991. Wetland Valuation: Guidelines and Techniques. Asian Wetland Bureau – Indonesia.

Jiang M, Lu XG, Xu LS, Chu LJ and Tong SZ. 2007. Flood mitigation benefit of wetland soil – A case study in Momoge National Nature Reserve, China. Ecological Economics 61: 217-223.

Johnston, C.A., Detenbeck, N.E & Niemi, G.J. 1990. The cumulative effect of wetlands on stream water quality and quantity. A landscape approach. Biogeochemistry 10: 105-141.

Johnston RJ, Grigalunas TA, Opaluch JJ, Mazzotta M and Diamantedes J. 2002. Valuing estuarine resource services using economic and ecological models: The Peonic Estuary System study. Coastal Management 30: 47-65.

Jones MGW. 2002. Developing a classification system for Western Cape wetlands. MSc. Thesis, University of Cape Town.

Jordan TE, Whigham DF, Hofmockel KH and Pittek MA. 2003. Nutrient and sediment removal by a restored wetland receiving agricultural runoff. Journal of Environmental Quality 32:1534-1547.

Jothiyangkoon C, Sivapalan M and Farmer L. 2001. Process controls of water balance variability in a large semi-arid catchment: downward approach to hydrological development, Journal of Hydrology 254: 174-198.

Kangalawe RYM and Liwenga ET. 2005. Livelihoods in the wetlands of the Kilombero Valley in Tanzania: Opportunities and challenges to integrated water resource management. Physics and Chemistry of the Earth 30: 968-975.

Kirschbaum, M.U.F. 2003 To sink or burn? A discussion of the potential contributions of forests to greenhouse gas balances through storing carbon or providing biofuels Biomass and Bioenergy 24: 297-310.

Kotze, D., and C. M. Breen. 1994. Agricultural land use impacts on wetland functional values. WRC Report No. 501/3/94.

Kotze DC, Marneweck GG, Batchelor AL, Lindley DS and Collins NB. 2008. WET-EcoServices: A technique for rapidly assessing ecosystem services supplied by wetlands. WRC Report no. TT 339/08, Water Research Commission, Pretoria, South Africa. 69pp.

Kramer RA and Sharma N. 1997. Tropical forest biodiversity protection: who pays and why. In: Last stand: protected areas and the defence of tropical biodiversity. Pp 162-186 in Kramer R, van Schaik C and Johnson J (eds.). Oxford University Press, New York, NY.

Krasnostein A.L. and Oldham C.E. 2004. Predicting wetland water storage. Water Resources Res. 40: W10203.

Kwak SJ, Yoo SH and Lee CK. 2007. Valuation of the Woopo Wetland in Korea: A contingent valuation study. Environment and Development Economics 12: 323-328.

Lamberth SJ and Turpie JK. 2003. The role of estuaries in South African fisheries: Economic importance and management implications. African Journal of Marine Science 25: 131-157.

Lange G-M, Hassan R and Hamilton K (eds.). 2003. Environmental Accounting in Action: Case Studies from Southern Africa. Edward Elgar Publishing, Cheltenham, UK. 223pp.

Lannas, K.S.M. and Turpie, J.K. 2009. Valuing the provisioning services of wetlands: contrasting a rural wetland in Lesotho with a peri-urban wetland in South Africa. Ecology and Society 14: 18 (online).

Leitch J and Hovde B. 1996. Empirical valuation of prairie potholes: five case studies. Great Plains Research 6: 25-39.

Limburg KE, O'Neil RV, Costanza R and Farber S. 2002. Complex systems and valuation. Ecological Economics 41: 409-20.

Lockwood M and Tracy K. 1995. Nonmarket economic valuation of an urban recreation park. Journal of Leisure Research 27: 155-167.

Lynne GD, Conroy P and Prochaska FJ. 1981. Economic valuation of marsh areas for marine production processes. Journal of Environmental Economics and Management 8:175-186.

Mafabi, P., Kakuru, W., Arinaitwe, J. and Kizito Y., 1998, Uganda's National Biodiversity Strategy and Action Plan: Wetlands Resources Subsector Component, National Environment Management Authority, Kampala.

Mahan B, Polasky S, Adams R. 2000. Valuing urban wetlands: A property price approach. Land. Economics 76: 100-113.

Matthews A. 1987. An economic valuation of the proposed *Clonmacnoise* heritage zone. Report for the Environmental Sciences Unit, Trinity College, Dublin. 100pp.

McKee, L.J., Eyre, B.D. and Hossain, S. 2000. Transport and retention of nitrogen and phosphorus in the sub-tropical Richmond River estuary, Australia—A budget approach. Biogeochemistry 50:241-278.

Millennium Ecosystem Assessment. 2003. Ecosystems and human well-being: A framework for assessment. Millennium Ecosystem Assessment. Island Press, Washington D.C. 245pp.

Ming, J., Xian-guo L, Lin-shu X, Li-juan C and Shouzheng T. 2007. Flood mitigation benefit of wetland soil — A case study in Momoge National Nature Reserve in China. Ecological Economics 61: 217-223

Mitchell RC and Carson TR. 1989. Using surveys to value public goods: The contingent valuation method. Resources for the Future, Washington D.C. 463pp.

Mitsch, W.J. and Gosselink, J.G., 1993. In: *Wetlands* (second ed.), Van Nostrand Reinhold, New York, 722 pp.

Mitsch WJ and Gosselink JG. 2000. The value of wetlands: Importance of scale and landscape setting. Ecological Economics 35: 25-33.

Moberg F and Folke C. 1999. Ecological goods and services of coral reef ecosystems. Ecological Economics 29(2): 215-233.

Naidoo R and Adamowicz LW. In press. Mapping opportunity costs of conservation in mixed natural/agricultural landscapes. Conservation Biology.

Napier V, Turpie JK and Clark B. 2009. Value and management of the subsistence fishery at Knysna estuary, South Africa. African Journal of Marine Science (in press).

Nhuan MT, Ninh NH, Huy LQ, Sam DD, Ha TH, Thanh NC, Oanh BK, Nga DT, Son NN and Du NQ. 2003. Vietnam wetland component report: Economic valuation of demonstration wetland sites in Vietnam. United Nations Environmental Programme, Global Environment Facility and UNEP/GEF South China Sea Project. Report number: Hanoi 4/2003.

Noble RG and Hemens J. 1978. Inland water ecosystems in South Africa – a review of research needs. South African National Scientific Programme Report 34. Pretoria, CSIR.

Norberg J. 1999. Linking nature's services to ecosystems: Some general ecological concepts. Ecological Economics 29(2): 183-202.

Novitzki, R.P. 1979. Hydrologic characteristics of Wisconsin 's wetlands and their influence on floods, stream flow, and sediment. p 377-388 In: Wetland functions and values: the state of our understanding. American Water Resources Association.

OECD. 1994. The economic appraisal of environmental projects and policies: A practical guide. OECD, Paris.

Ogawa, H. and Male, J.W. 1986. Simulating the flood mitigation role of wetlands.Journal of Water Resources Planning and Management. 112: 114-128.

Oglethorpe DR and Miliadou D. 2000. Economic attributes of the non-use attributes of a wetland: A case study for Lake Kerkini. Journal of Environmental Planning and Management. 43: 755-767.

Osano P, Balmford A, Rouget M, Turpie J and Thuiller W. Submitted. Estimating land prices and opportunity costs of conservation in a mega-diversity country.

Patrick WH and Mahapatra IC. 1968. Transformation and Availability to Rice of Nitrogen and Phosphorus in Waterlogged Soils. Advances in Agronomy 20: 323-359.

Palmer RW, Turpie J, Marnewick GC and Batchelor AL. 2002. Ecological and economic evaluation of wetlands in the upper Olifants River catchment, South Africa. WRC Report No. 1162/1/02. 138pp.

Pearce D, Whittington D, Georgiou S and Moran D. 1994. Economic values and the environment in the developing world: a report to the United Nations Environment Programme Nairobi. Environmental Economics Series, Paper No. 14, United Nations Environmental Programme, Environment and Economics Unit.

Pearce DW and Turner RK. 1990. Economics of Natural Resources and the Environment. Harvester Wheatsheaf, Hertfordshire. 280pp.

Peltier, E.F., Webb, S.M. and Gaillard, J.-F. 2003. Zinc and lead sequestration in an impacted wetland system. Advances in Environmental Research 8: 103-112.

Perman R, Ma Y and McGilvray J. 1996. Natural Resource and Environmental Economics. Longman, London and New York. 396pp.

Pitcher, T.J. and Preikshot, D.B. 2001. Rapfish: A Rapid Appraisal Technique to Evaluate the. Sustainability Status of Fisheries. Fisheries Research 49(3): 255-270.

Plantec Africa, MTK Planning Solutions and Lesedi Consulting, 2006. *Okavango Delta RAMSAR Site land use and land management plan (2005-2029)*. Prepared for the Tawana land Board.

Ponnamperuma, F.N. 1972. The Chemistry of Submerged Soils. Advances in Agronomy 24: 29-96.

Ramsar. 1971. Convention on wetlands of international importance especially as waterfowl habitat. UN Treaty Series No. 14583.

Reynolds JE and Regalado A. 2002. The effects of wetlands and other factors on rural land values. Appraisal Journal 72:182-190.

Richardson, C.J. 1985. Mechanisms controlling phosphorus retention capacity in freshwater wetlands. Science 228: 1424-1427.

Richardson, C.J. & Craft, CB. Effective phosphorous control in wetlands: fact or fiction? PP 271-292. In: Moshiri, G.A. Constructed wetlands for water quality improvement. CRC Press.

Rogers KH. 1995. Riparian wetlands. In: Wetlands of South Africa. Pp 41-52 in Cowan GI (ed.). Department of Environmental Affairs and Tourism, Pretoria, South Africa.

Rogers F.E., Rogers K.H. and Buzer J.S. 1985. Wetlands for wastewater treatment: with special reference to municipal wastewaters. WITS University Press, Johannesburg.

Rönnbäck P and Primavera JH. 2000. Illuminating the need for ecological knowledge in economic valuation of mangroves under different management regimes – a critique. Ecological Economics 35: 135-141.

Rozan A. 2004. Benefit transfer: a comparison of WTP for air quality between France and Germany. Environmental and Resource Economics 29: 295-306.

Ruitenbeek HJ. 1992. Mangrove management: An economic analysis of management options with a focus on Bintuni Bay, Irian Jaya. Environmental Management in Indonesia Project Environmental Reports. Report No. 8. Jakarta and Halifax.

Ruitenbeek HJ. 1994. Modelling economy-ecology linkages in mangroves: Economic evidence for promoting conservation in Bintuni Bay, Indonesia. Ecological Economics 10: 233-247.

Saunders, D.L. and Kalff, J. 2001. Nitrogen retention in wetlands, lakes and rivers. Hydrobiologia 443: 205-212.

Schwabe CA. 1995. Alpine mires of the eastern highlands of Lesotho. In: Wetlands of South Africa. Cowan GI (ed.). Department of Environmental Affairs and Tourism, Pretoria, South Africa. 33-40. 290pp.

Schuyt KD. 2005. Economic consequences of wetland degradation for local populations in Africa. Ecological Economics 53: 177-190.

San José JJ and Montes RA. 2001. Management effects on carbon stocks and fluxes across the Orinoco savannas. Forest Ecology and Management 150: 293-311.

Sathirathai S. 1997. Economic valuation of mangroves and the roles of local communities in the conservation of natural resources: case study of Surat Thani, south of Thailand. Economy and Environment Programme for Southeast Asia Research Report.

Shultz S and Taff S. 2004. Calculating wetland easement payments with alternative land value data: A case study of the USFW small wetland acquisition program in the prairie pothole region. Journal of Soil & Water Conservation 59(3):103-109.

Seidl AF and Moraes AS. 2000. Global valuation of ecosystem services: Application to the Pantanal da Nhecolandia, Brazil. Ecological Economics 33: 1-6.

Seyam IM, Hoekstra AY, Ngabirano GS and Savenije HHG. 2001. The value of freshwater wetlands in the Zambezi basin. Value of Water Research Report Series No. 7.

Shaw WD. 1992. Searching for the opportunity cost of an individual's time. Land Economics 68: 107-115.

Smakhtin, V.U. & Batchelor, A.L. 2005. Evaluating wetland flow regulating functions using discharge time-series. *Hydrological processes* 19: 1293-1305.

Shrestha R, Loomis J. 2001. Testing a meta-analysis model for benefit transfer in international outdoor recreation. Ecological Economics 39: 67-84.

Shrestha RK, Seidl AF and Moraes AS. 2002. Value of recreational fishing in the Brazilian Pantanal: A travel cost analysis using count data models. Ecological Economics 42: 289-299.

Siegel, D.I. 1988. The recharge-discharge function of wetlands near Juneau, Alaska: Part I. Hydrological investigations. Groundwater 26: 427-433.

SIWI 2004. Making water a part of economic development. Studies in Methods Series F No. 61, Statistics Division, United Nations Department of Economic and Social Affairs, United Nations, New York, European Commission, Brussels, International Monetary Fund, Washington, DC, Organisation for Economic Co-operation and Development, Paris, and World Bank, Washington DC. 572pp

Smith VK, Desvousges WH and McGirney MP. 1983. The opportunity cost of travel time in recreation demand models. Land Economics 59: 259-278.

Söderqvist T, Mitsh WJ and Turner RK. 2000. Valuation of wetlands in a landscape and institutional Perspective. Ecological Economics 35: 1-6.

Spaninks F and van Beukering P. 1997. Economic valuation of mangrove ecosystems: Potential and limitations. CREED Working Paper No. 14. IIED and IVM-VU.

Sterner, T. 2007. Gasoline taxes a useful instrument for climate policy. Energy Policy 35(6): 3194-3202.

Stevens TH, Belkener R, Dennis D, Kittredge D and Willis C. 2000. Comparison of contingent valuation and conjoint analysis in ecosystem management. Ecological Economics 32: 63-74.

Stewart TJ, Scott L and Iloni K. 1993. Scenario based multicriteria policy planning for water management in South Africa. WRC Report No 296/1/93.

Streever WJ, Callghan-Perry M, Searles A, Stevens T and Svoboda P. 1998. Public attitudes and values for wetland conservation in New South Wales, Australia. Journal of Environmental Management 54: 1-14.

Stuip MAM, Baker CJ and Oosterberg W. 2002. The Socio-economics of Wetlands. Wetlands International and RIZA, Wageningen, The Netherlands. 35pp.

Thompson JR and Hollis GE. 1995. Hydrological Modelling and the Sustainable Development of the Hadejia-Nguru Wetlands, Nigeria. Hydrological Sciences Journal 40 1, 97-116.

Thullen, J.S., Sartoris, J.J. and Nelson, S.M. 2005. Managing vegetation in surface-flow wastewater-treatment wetlands for optimal treatment performance Ecological Engineering 25: 583-593.

Tilton, D.L. and Kadlec, R.H. 1979. The utilization of a fresh-water wetland for nutrient removal from secondarily treated waste water effluent. Journal of Environmental Quality 8:328-334.

Tong CF, Feagin RA, Lu JJ, Zhang XF, Zhu XJ, Wang W and He WS. 2007. Ecosystem services values and restoration in the urban Snyang wetland of Wenzhou, China. Ecological Engineering 29: 249-258.

Turner K. 1991. Economics and wetland management. Ambio 20: 59-63.

Turner RK, Folke C, Gren IM, and Bateman JI. 1997. Wetland Valuation: Three case Studies. Pp129-149. In: Perrings C, Maler KG, Folke C, Holling CS and Jansson BO (eds.). Biodiversity Loss: Economic and Ecological Issues. Cambridge University Press. Cambridge.

Turner K, Paavola J, Cooper P, Farber S, Jessamy V and Georgiou S. 2003. Valuing nature: Lessons learned and future research directions. Ecological Economics 46: 493-510.

Turner RK, van den Bergh JCJM, Söderqvist T, Barendregt A, van der Straaten J, Maltby E and van Ierland EC. 2000. Special issue: The values of wetlands: Landscape and institutional perspectives. Ecological-economic analysis of wetlands: Scientific integration for management and policy. Ecological Economics 35: 7-23.

Turpie JK. 2000. The use and value of natural resources of the Rufiji Floodplain and Delta, Tanzania. Rufiji Environmental Management Project and IUCN – Eastern Africa Regional Office. Ecological and Resource Economics Consulting. 116pp.

Turpie JK. 2003. The existence value of biodiversity in South Africa: How interest, experience, knowledge, income and perceived level of threat influence local willingness to pay. Ecological Economics 46: 199-216.

Turpie JK. 2006. Kromme/Seekoei Catchments Reserve Determination Study – Technical Component. Socio-economics Report. Prepared by Anchor Environmental Consultants for Coastal and Environmental Services. Report no. RDM/ K90/ 00/CON/1105. 91pp.

Turpie JK and Barnes JI. 2003. Balancing ecology and economics in the multiple use of wetlands on communal lands. Proceedings VIIth International Rangelands Congress, Durban, South Africa, 26 July-1 August 2003. 1932-1941.

Turpie J, Barnes J, Arntzen J, Nherera B, Lange G and Buzwani B. 2006b. Economic value of the Okavango Delta, Botswana, and implications for management. Okavango Delta Management Plan. 139pp.

Turpie JK and Clark BM. 2007. The health status, conservation importance, and economic value of temperate South African estuaries and development of a regional conservation plan. Report to CapeNature

Turpie JK and Egoh B. 2003. Contribution of natural resources to rural livelihoods around Lake Liambezi and Bukalo Channel, eastern Caprivi and impacts of proposed agricultural developments and artificial recharge of the lake. Report to Afridev, for the Government of Namibia. 89pp.

Turpie JK and Joubert AR. 2001. Estimating potential impacts of a change in river quality on the tourism value of Kruger National Park: An application of travel cost, contingent and conjoint valuation methods. Water SA 27: 387-398.

Turpie JK and Joubert AJ. 2004. Estimating the recreational use value of estuaries: A case study of the Knysna estuary. In: Proceedings of a national workshop on resource economics as a tool for the management and conservation of estuaries in South Africa. Turpie JK and Hosking S (eds.).   Report by Anchor Environmental Consultants for Department of Environmental Affairs and Tourism. 120pp.

Turpie J, Joubert A, van Zyl H, Harding B and Leiman A. 2001. Valuation of open space in the Cape Metropolitan Area. A pilot study to demonstrate the application of environmental and resource economics methods for the assessment of open space values in two case study areas: Metro South and Metro South-east. Report to the City of Cape Town. 104pp.

Turpie JK, Scovronick N and Brown C. 2006b. Intangible resource losses and gains in Phase 1 areas of the Lesotho Highlands Water Project. Report to LHDA. 69pp.

Turpie JK, Sihlope N, Carter A, Maswime T and Hosking S. 2006a. Maximising the socio-economic benefits of estuaries through integrated planning and management: A rationale and protocol for incorporating and enhancing estuary values in planning and management. Report to the Water Research Commission. 117pp.

Turpie J, Smith B, Emerton L and Barnes J. 1999. Economic value of the Zambezi basin wetlands. Report to IUCN. 199pp.

Turpie, J.K. and van Zyl, H. 2002. Valuing the environment in water resources management. Pp 85-110, in: Hirji, R., Johnson, P., Maro, P. & Matiza Chiuta, T. (eds) Defining and mainstreaming environmental sustainability in water resources management in southern Africa. SADC, IUCN, SARDC, World Bank: Maseru/Harare/Washington DC.

Turpie, J.K., Winkler, H. & Midgley, G. 2004. Economic impacts of climate change in South Africa: a preliminary analysis of unmitigated damage costs. Pp. 130-160 in: Blignaut, J.N. & De Wit, M.P. Sustainable Options. UCT Press.

Van Zyl H and Leiman A. 2002. Hedonic approaches to estimating the impacts of open spaces: A case study in the Cape. South African Journal of Economics and Management Science 5: 379-394.

Vellidis, G Lowrance, R Gay, P and Hubbard, R.K. 2003. Nutrient transport in a restored riparian wetland. Journal of Environmental Quality 32:711-726

Verhoeven JTA, Arheimer B, Yin C and Hefting MM. 2006. Regional and global concerns over wetlands and water quality. Trends in Ecology and Evolution 21: 96-103.

Viglizzo, E.F. and Frank. F.C. 2006. Land-use options for Del Plata Basin in South America: Tradeoffs analysis based on ecosystem service provision. Ecological Economics 57: 140-151.

Vorhies F. 1999. Environmental economics explained. IUCN. Accessed online at [http://biodiversityeconomics.org] on 3 August 2007.

Welcomme, R.L., 1985. River fisheries. FAO Tech. Pap., 262: 330 pp.

Willis K and Garrod GD. 1991a. An individual travel-cost method of evaluating forest recreation. Journal of Agricultural Economics 42: 33-42.

Willis K and Garrod GD. 1991b. Landscape values: A contingent valuation approach and case study of the Yorkshire Dales National Park. Countryside Change Working paper 21, University of Newcastle upon Tyne. 36pp.

Wilson CG, Matisoff G, Whiting PJ and Klarer DM. 2005. Transport of fine sediment through a wetland using radionuclide tracers: Old Woman Creek, OH. Journal of Great Lakes Research 31(1): 56-67.

Wilson MA and Carpenter SR. 1999. Economic valuation of freshwater ecosystems services in the United States 1971-1997. Ecological Applications 9: 772-83.

Wilson MA and Howarth RB. 2002. Valuation techniques for achieving social fairness in the distribution of ecosystem services. Ecological Economics 41: 431-43.

Witt R. 006. Food security and poverty alleviation through improved valuation and governance of river fisheries in Africa. Research proposal. University of Hannover, Germany.

Woltemade, C. J. and K. W. Potter. A watershed modeling analysis of fluvial geomorphologic influences on flood peak attenuation. Water Resources Research 30(6): 1933-1942.

Woodward RT and Wui Y. 2001. Analysis: The economic value of wetland services: A meta-analysis. Ecological Economics 37: 257-270.

World Bank. 1998. Economic Analysis and environmental assessment. Accessed online at [http://www.worldbank.org/wbi/sourcebook/sba104.htm] on 3 August 2007.

Young LJ, Roeder E, Ursin E. 2008. Nitrogen Input Assessment for the Wekiva Study Area. An Educational Poster for: The Florida Department of Health by: University of Florida, Institute for Food and Agricultural Sciences.

GLOSSARY

Allocate – to award a certain quantity of a 'resource' (such as land or water) to various users or to different uses.

Carbon sequestration – the process of capturing carbon and keeping it from entering the atmosphere for some period. Carbon is sequestered in carbon sinks, such as forests, soils or oceans.

Carrying capacity – a biological term that indicates the ability or capacity of an area to support or 'carry' plant and animal life. In human terms it is the number of people that can be supported by an area.

Consumer surplus – a net benefit realised by consumers when they buy a good at the prevailing market price. It is the difference between the maximum price consumers would be willing to pay and that which they actually pay for the units of the good purchased.

Contingent valuation – the use of questionnaires about valuation to estimate the willingness of respondents to pay for public projects or programmes

Direct use value – within the 'total economic value framework', the benefits derived from the goods and services provided by an 'ecosystem' that are used directly by an economic agent. These include consumptive uses (e.g. harvesting goods) and non-consumptive uses (e.g. enjoyment of scenic beauty).

Discount rate – the interest rate at which future payments or income are discounted in a multi-period model. Reflects the time preference between consumption and income now or in the future.

Discounting – the process of applying a 'discount rate'. The rate of interest to cost and benefit flows that is used to find the equivalent value today of sums receivable or payable in the future.

Economic growth – the percentage change in the 'national income', resulting from investment, increases in trade, size or scale effects, or technological progress.

Ecosystem services – the benefits people obtain from 'ecosystems', including provisioning of food and water, regulation of disease and flooding, spiritual, recreational and cultural benefits.

Existence value – the value that individuals may attach to the mere knowledge of the existence of something, as opposed to having direct use thereof. Part of non-use value.

Flow accounts – Used here to refer to production accounts in 'natural resource accounts', valued in terms of annual contribution to national income.

Gross domestic product (GDP) – the measure of total 'value added' (total value of all the goods and services produced in an economy, less raw materials, and other goods and services used in the production process) in all resident producing units, during some accounting period, usually a year. See 'national income'.

Gross income – 'Gross revenue', 'turnover', usually a private measure.

Gross national product (GNP) – the same as GDP except that it includes income earned abroad by nationals, and excludes income transferred abroad by foreign owners. See 'national income'.

Gross output – 'gross revenue' in economic terms, commonly the aggregate of all gross revenues in the economy.

Gross revenue – in general terms, equal to the unit price multiplied by the quantity of units sold by a production unit. Here used as a private value.

Indirect use value – the benefits derived from the goods and services provided by an ecosystem that are used indirectly by an economic agent. For example, an agent at some distance from an ecosystem may derive benefits from drinking water that has been purified as it passed through the ecosystem.

Marginal value – change in economic value associated with a unit change in output, consumption or some other economic choice variable.

Molapo – a grass-covered depression that fills with water during the wet season. Also called a '*dambo'*.

National accounts – the compilation of accounts to derive estimates of the 'national income'.

National income – the total net earnings of labour and property employed in the production of goods and services in a nation during some accounting period, usually a year. Commonly measured by the 'gross domestic product' (GDP) the 'gross national product' (GNP), and the 'gross national income' (GNI). Measured either as the value of all expenditure on final goods

and services, the value of all payments to factors of production, or the value of all value added by producing units.

Natural asset value – capital value of the stock of a natural resource. This is the present value of the stream of future 'economic rents' ('resource rents') that a natural resource will generate. Present values are typically obtained by 'discounting' future benefits and costs.

Natural resource accounts – the compilation of asset and 'flow accounts' for natural assets, to complement the 'national accounts'. Asset accounts are valued in terms of 'natural asset value', flow accounts are valued in terms of 'national income'.

Net income – 'profit', a private value.

Net national income – 'Gross national income' less depreciation of assets.

Net present value – the present value of an investment, found by 'discounting' all current and future streams of income or expenditure by a 'discount rate'.

Non-use value – see 'existence value'

Open access resource – a good or service over which no property rights are recognised.

Opportunity cost – the benefits foregone by undertaking one activity instead of another.

Option value – the value of preserving the option to use services in the future.

Resource rent or **economic rent** – the return a factor of production receives in excess of the minimum required to bring forth the service of the factor, or the surplus available in a 'production unit' after accounting for the costs of production including a reasonable return to capital. Resource rent is the economic rent generated from use of a natural resource.

Social accounting matrix (SAM) – an economic input-output model of the national economy, used as a tool for impact analysis. Expands the national accounts to show the linkages between production and generation of income and distribution of income

Social costs and benefits – costs and benefits as seen from the perspective of society as a whole. These differ from private costs and benefits in being more inclusive (all costs and benefits borne by some member of society are taken into account) and in being valued at social opportunity cost rather than market prices, where these differ. Sometimes termed 'economic' costs and benefits.

Sustainable – something that can carry on indefinitely.

Sustainable development – development that can support people now and carry on supporting people for a long time into the future, without having effects that threaten the livelihoods of future generations

Total economic value framework – a widely used framework to disaggregate the components of utilitarian value, including 'direct' and 'indirect use value', 'option value' and 'existence value'. Commonly applied to natural resources.

Turnover – 'gross revenue', 'gross income'.

Value added – the amount of economic value generated by the activity carried on within each 'production unit' in the economy, the difference between the 'gross revenue' of the production unit and the inputs purchased from outside the production unit. When aggregated for the whole economy becomes a measure of 'national income'.

APPENDIX 1

Table A1: Examples of values obtained in wetland valuation studies

Wetland location	Size	Type of value	Value	Value in US dollars	Year (pub.)	Reference
Africa						
Nakivubo urban wetland, Uganda	5.29 km² (529 ha)	Crop cultivation	USh 199.34 m/y	73 420/y	1998	Emerton et al., 1998
		Papyrus harvesting	USh 17.41 m/y	11 607/y		
		Brick making	USh 32.00 m/y	213 333/y		
		Fish farming	USh 6.07 m/y	4 047/y		
		Water treatment	USh 1 247.8-2 314.0 m/y	831 853-1 542 687/y		
		Total wetland value	USh 1 763.5-2 479.6 m/y	1 175 660-1 653 093/y		
Uganda's wetlands	Over 30 000 km² (> 3 mha)	Papyrus utilisation	USh 5 923 m/y	4 379 298/y	1998	Mafabi et al., 1998
		Livestock	USh 18 114 m/y	13 392 976/y		
		Agricultural production	USh 65 962 m/y	48 770 425/y		
		Water purification	USh 5 300 m/y	3 918 669/y		
Zambezi Basin wetlands	Barotse 550 000 ha	Indirect use value (net present value)			1998	Turpie et al., 1999
		Barotse		43.9 m		
	Chobe-Caprivi 304 600 ha	Chobe – Caprivi		22 m		
		Lower Shire		36.6 m		
		Zambezi Delta		79.9 m		
	Lower Shire 243 000 ha	Consumptive use value (Net economic value/y)				
		Barotse		8 647 000/y		
	Zambezi Delta 1 789 000 ha	Chobe – Caprivi		4 770 000/y		

Wetland location	Size	Type of value	Value	Value in US dollars	Year (pub.)	Reference
		Lower Shire		19 854 000/y		
		Zambezi Delta		11 747 000/y		
		Existence value (Present value of WTP) for				
		Barotse		4 229 309		
		All Zambian wetlands		16 651 506		
Rufiji floodplain and delta, Tanzania	Rivers and lakes – 42 531 ha	Harvesting resources			2000	Turpie, 2000
		Rivers and lakes		42/ha/y		
		Floodplain		2/ha/y		
		Mangroves		17/ha/y		
	Floodplain – 179 599 ha	Inputs to agriculture				
		Floodplain		65/ha/y		
		Nursery function				
	Mangroves – 55 154 ha	Mangroves		41/ha/y		
		Carbon sequestration				
		Floodplain		15/ha/y		
		Mangroves		650/ha/y		
Cape Town Metropolitan wetlands, South Africa	Total wetland area not given	Recreational value of metropolitan wetlands (WTP)	R1900-4300/ha/y	220-500/ha/y	2001	Turpie et al., 2001
		Recreational value of metropolitan wetlands (residents – property price premium)				
	Sandvlei: 155 ha	Recreational value of Sandvlei (visitors	R406 000/ha	47 209/ha		
		Recreational value of				

Wetland location	Size	Type of value	Value	Value in US dollars	Year (pub.)	Reference
		Sandvlei (residents – property value premium)	R700 000/y	81 395/y (525/ha/y)		
		Water storage and purification function (annualised replacement cost)	R84.25 m (R543 500/ha)	9.80 m (63 200/ha)		
			R18-20 000/ha/y	2 100-2 325/ha/y		
Olifants River wetlands, Mpumalanga, South Africa	Not given	Direct use value			2001	Palmer et al., 2002
		Riparian wetlands	R12-102/ha/y	1.4-12/ha/y		
		Seepage wetlands	R1800-2500/ha/y	209-290/ha/y		
		Pans	R3150-3250/ha/y	366-378/ha/y		
		Artificial wetlands	R1750-1850/ha/y	203-215/ha/y		
Lake Chilwa wetland, Malawi	2400 km² (240 000 ha)	Agriculture		1.2 m/y	2002	Schuyt, 2005
		Fish		18.7 m/y		
		Vegetation and clay		14 000/y		
		Water transport		436 000/y		
		Grasslands (grazing etc.)		638 000/y		
Hadejia-Nguru wetlands, Nigeria	3 500 km² (350 000 ha)	Groundwater recharge		17 391/y	2002	Schuyt, 2005
		Agricultural activities		11 m/y		
		Fishing		3.5 m/y		
		Fuel wood		1.6 m/y		
		Doum palm		130 000/y		
		Potash		900/y		

Wetland location	Size	Type of value	Value	Value in US dollars	Year (pub.)	Reference
Okavango Delta, Botswana	13 000 km^2 (1 300 000 ha)	Net value to households Livestock production Crop production Natural resources Total net value to hh	Pula 1.2 m/y 0.9 m/y 14.2 m/y 16.35 m/y (16.51 m/y value added)	0.22 m/y 0.16 m/y 2.63 m/y 3.03 m/y (3.06 m/y)	2006	Turpie et al., 2006b
		Tourism value	P1115 m turnover, P401 m/y value added	74.3 m/y value added		
		Indirect use values Groundwater recharge Carbon sequestration Wildlife refuge Water purification Education and scientific value	P16 m/y P86 m/y P77 m/y P2.2 m/y P18 m/y	2.96 m/y 15.92 m/y 14.23 m/y 0.41 m/y 3.33 m/y		
Yala wetland, Kenya	17 500 ha	Total annual fish yield	Ksh 11 million		2006	Abila and Othina (2006)
Asia						
Bintuni Bay mangrove, Indonesia	300 000 ha	Total value of household income from marketed and non-marketed sources	Indonesian Rupiah 9 m/hh/y	4 500/hh/y	1991	Barbier, 1993
Kuantu wetland, Taiwan	153 ha	Existence value (annual WTP values into perpetuity, discounted at 5-10%)		200 million to 1.2 billion	(2001)	Hamitt et al., 2001
Coastal Thailand	Not given	Expected output from shrimp farming and sustainable		160-500/ha/yr	(1991)	Turner, 1991

Wetland location	Size	Type of value	Value	Value in US dollars	Year (pub.)	Reference
		yield forestry				
Sanyang wetland, China	11.41 km² (1141 ha)	Current value Potential value	5807 yuan/ha/y 55 332 yuan/ha/y	745/ha/y 7 096/ha/yr	(2007)	Tong et al., 2007
Woopo wetland, Korea	854 ha	Average willingness to pay (conservation value)	2 731 to 3 960/hh/y	2.10-3.05/hh/yr	(2007)	Kwak et al., 2007
Momoge National Nature Reserve, China	14.4 x 104 ha, 80% of which is wetland	Flood mitigation benefit of wetland soil		5 700/ha/yr	2004	Jiang et al., 2007
Europe						
Martebo mire, Sweden	Not given	Existence value in terms of WTP for replacing lost wetland goods and services		0.4-1.2 m/y	1989	Turner et al., 1997
Broadland, United Kingdom	Not given	Mean dichotomous choice willingness to pay	£244/hh/y	456/hh/y	1991	Turner et al., 1997
Gotland, Sweden	Not given	Value of an increase in wetland nitrogen abatement due to restoration	859 Swedish Kroner/kg/yr	148/kg/yr	(1997)	Turner et al., 1997
De Wieden wetlands, Netherlands	5 200 ha	Total value of select ecosystem services	Euros/year 4 500 000	5 935 500/yr	(2006)	Hein et al., 2006
Lake Kerkini, Greece	5000-7500 ha, depending on season	Existence value: Aggregate mean annual WTP for protection of the lake by Macedonians	Gdr 12 800/y	35 389 226/y	(2000)	Oglethorpe and Miliadou, 2000
Australia						

Wetland location	Size	Type of value	Value	Value in US dollars	Year (pub.)	Reference
New South Wales wetlands	Not given	Aggregate willingness to pay for wetland conservation	A$38 million per year for 5yrs	23 322 4999/yr for 5yrs	(1998)	Streever et al., 1998
North America						
Louisiana coastal wetlands study area, USA	1.32 million ha	Average gross economic value of wetlands based recreation, 1986-1987		110.03/ha/yr	1986-1987	Bergstrom et al., 1990
Natural coastal wetlands, Louisiana, USA	Not given	Total present value of an average acre (at 3%-8% discount rate)		2 429-17 000 acre	1983	Costanza et al., 1989
Five prairie potholes, North Dakota, USA	890 ha of wetland plus 970 ha of associated uplands	Average annual values per hectare: User values Owner values Regional activity Social values		43.8/ha/y 29.4/ha/y 465/ha/y 73.2/ha/y	1993	Leitch and Hovde, 1996
South America						
Pantanal	138 000 km² (13 800 000 ha)	Gross annual value of: Ecosystem services Agriculture		5726.9 ha/y 23.5 ha/y	(2006)	Viglizzo and Frank. 2006
Brazilian Pantanal	138 000 km² (13 800 000 ha)	Consumer surplus values of recreational fishing Social welfare estimate		540.54-869.57/trip 35-56 m	1994	Shrestha et al., 2002
Pantanal da Nhecolandia, Brazil	26 921 km² (2 692 100 ha)	Total annual value		5811.11/ha/y	1994	Seidl and Moraes,2000