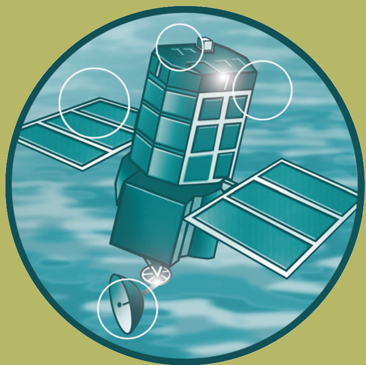
Joint Defra/EA Flood and Coastal Erosion
Risk Management R&D Programme

Development of economic appraisal methods for flood management and coastal erosion protection

Development of tools for the economic valuation of multi-functional wetlands: methods and techniques

R&D Technical Report FD2014/TR3



Environment Agency

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Development of Economic Appraisal Methods for Flood Management and Coastal Erosion Protection

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Statement of Use
This report describes tools for the multi functional economic valuation of wetlands and makes recommendation for their use in the evaluation of flood risk management options.

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FOREWORD

This report was produced by The Centre for Social and Economic Research at the University of East Anglia on the Global Environment (CSERGE), under the Policy Development theme of the joint Defra and Environment Agency R&D Programme. The final text is the result of collaboration between the Centre, the Flood Hazard Research Centre, Defra and the Agency.

The aim of this report was to advise on the valuation of wetland functions and make initial recommendations for methods to use in the appraisal of flood risk management options. To this aim the report offers:

- a wide review of the rationale for valuing wetland functions in both policy and project appraisal;

- a helpful reference for: Defra when considering policy; and operating authorities when appraising flood risk and coast erosion management options;

- how legal obligations and high-level Government targets should be taken account in valuation exercises, thus giving important policy context to the appraisal process;

- how it is not only the wetland that should be valued but also the goods and services provided, which affect human welfare. Although concentrating on wetlands, the approaches suggested may provide an assessment framework for other types of environmental asset.

Importantly, the report: provides approaches that complement the policies set out in the Treasury 'Green Book'; and offers a set of recommendations for policy development and for appraisal advice, in line with the emerging policy agenda and the Government Strategy, 'Making Space for Water'[1]. The report is also linked to other deliverables within the Joint R&D Programme, such as Multi Criteria Analysis development, and is referred to in the environment chapter of Middlesex University's new Multi Coloured Handbook.

Formal policy guidance referring to the evaluation techniques within the report will be developed in the future. Please look out for further information, at: http://www.defra.gov.uk/environ/fcd/default.htm. The Agency may also develop the findings of this research and take account of environmental costs and benefits within their operational guidance.

Defra thanks Professor Kerry Turner and his team for this report. We hope you find the report useful.

Defra Flood Management Division
October 2005

[1] see: http://defraweb/environ/fcd/policy/strategy/1stres.pdf

EXECUTIVE SUMMARY

Wetland ecosystems account for about 6% of the global land area and are among the most at risk of all environmental resources. The wetlands found in temperate climate zones such as the UK have long suffered significant losses and continue to face threats from, industrial, agricultural and residential developments, as well as from pollution and climate change.

Wetlands are complex ecological systems whose structure and processes provide society with valuable goods involving some direct utilisation of one or more wetland characteristics (known as direct use value). They also provide ecologically related services, supporting or protecting human activities or assets without being used directly (known as indirect use value). Wetland systems, as well as their distinctive landscapes, are also often significant socio-cultural assets (some of which are non-use values). So, the stock of wetlands is a multifunctional resource generating substantial socio-economic values (an aggregation of which is expressed as total economic value). A number of pieces of legislation (including international Convention Agreements) have been passed seeking to address in some way or another the wetland loss/degradation problem and the policy responses in terms of a sustainable management strategy. The core objective of this report is to examine and make initial recommendations on a decision support system that could underpin the sustainability strategy. In particular, the focus is on the incorporation of economic valuation methods and techniques into wetland assessment (i.e. conversion, restoration or creation). In other words to quantify the role that wetlands can play in catchment strategies dealing with, for example, flood alleviation, water quality protection/enhancement and biodiversity conservation. The methods and techniques advocated in this report are consistent with existing Treasury and other official project (DEFRA PAG 3) policy and programme assessment guidelines. They are relevant to flood protection (fluvial and saline flooding risks), sea defence and coastal protection appraisals and the emerging task of river basin management as set out in the EU Water Framework Directive.

It is recommended that the appraisal process be set within an analytical framework based on the so-called ecosystem function approach, allied to an extended cost-benefit appraisal (with an inherent identification of stakeholders, gainers and losers). This functional perspective lays down a four stage sequence through which the analysis proceeds. The initial stage requires a working classification of all relevant wetland types. It is not necessary, for assessment purposes, to construct a detailed and complex wetland classification system. A simple typology linked to a set of wetland functions is a sufficient basis for the valuation of most wetlands. Thus the second stage in the ecosystem function approach is the determination of the set of functions provided by particular wetlands e.g. flood water detention, nutrient retention, carbon sequestration, habitat provision etc.

The third stage involves the matching of functions and/or combinations of functions with outcomes in terms of tangible and intangible goods and services utilised or appreciated by humans (i.e. welfare affecting). The set of functions and related goods/services that need to be valued will be influenced by the policy content in which the appraisal is taking place e.g. total wetland loss, wetland degradation and partial loss of functionality, wetland creation, wetland trade-offs. Some dimensions of the value society may place on wetlands will remain outside of any monetary calculus. It is the case that in the UK and elsewhere legislation and designation measures are already in place to protect environmental assets and many of the more intangible values they possess. Options for the extension and/or reorientation of such standards, zones or practices will need to be assessed from a cost-effectiveness viewpoint i.e. changes must be costed in order to determine the least-cost option(s) available for implementation.

The final stage is concerned with the monetary valuation of the goods and services provided by the wetlands. While the economic valuation of a range of wetland goods and services is a practicable and meaningful exercise, the limits to this monetising approach should also be borne in mind, especially in terms of transferring economic value data across time and geographical space; and the existence of intangible natural values of cultural, historical and even ethical significance (combinations of these characteristics help to determine the social significance or rarity/scarcity value sometimes assigned to environmental assets and are accounted for by conservation designation and legislation).

The type of economic valuation method applied depends on the function(s) being valued and the type of environmental change that is the focus of policy attention. Pragmatically, the choice of technique and method will often be reduced to the valuation of one, or a small number, of functions (with due regard to the avoidance of double counting) as a "representative" value of the whole wetland. Alternatively, the estimation of a single overall composite value, via a survey method like contingent valuation, to serve as the proxy for the individual function values within the wetland is another approach.

A number of economic valuation techniques can be deployed if individual or a small number of functions are being valued. These include market pricing and productivity changes, implicit pricing derived from the analysis of goods for which markets exist and which incorporate particular environmental characteristics (hedonic pricing), travel cost estimates for recreation value and damage costs avoided if a wetland provides flood protection. But if an overall aggregate value for any given complex wetland is required then only a survey-based contingent valuation, or choice experiment, method will be appropriate.

A "whole landscape" catchment scale should be adopted to account for the fact that similar wetland classes may provide different mixes of "valued" goods/services depending on their spatial location in a catchment. These locational factors – e.g. proximity of wetland to human settlement, land use surrounding the wetland, existence of nature conservation schemes or zones within or nearby the wetland etc. – should be an integral component of any assessment.

Because of the interrelationships between functions (complementary and competitive) the double counting of benefits provided is an ever present danger. The avoidance of double counting requires a full appreciation of the linkages between wetland function, the provision of goods/services and a change in human welfare. It is only the last link i.e. the change in human welfare that is the focus of value (gain or loss). Thus, for example, in the case of an estuarine wetland (existing or created) one of its functions is the retention of nutrients via sedimentation storage. Because of this “natural” storage service water quality is enhanced and the public may recognise an amenity benefit. It is not correct, however, to add a value estimate for the nutrient storage service (e.g. the equivalent saving in conventional sewage/water treatment costs) to a value estimate representing the public’s amenity gain from improved water quality. This is double counting as only the amenity gain should be included in the valuation exercise.

A costing procedure that considers the savings in sewage/water treatment costs would be a valid approach if the estuary concerned was subject to, for example, existing water quality standards or targets and the policy question was how best to attain the target/standard. In these circumstances the aim would be to determine the “least-cost” (most “cost-effective”) set of measures necessary to meet the quality target/standard (the benefits of which have already been decided on behalf of society by the legislators). An abatement strategy, including different combinations of sewage treatment provision, nitrate sensitive zoning in catchments and intertidal wetland nutrient storage creation could then be assessed in cost-effectiveness terms i.e. what is the least cost combination of measures sufficient to meet the quality targets/standard provisions?

In conclusion, it is argued that the economic (monetary) valuation of a range of wetland goods/services is a practicable and meaningful exercise. That a typology of values based on the total economic value concept is an appropriate way to represent the long run and multi-faceted nature of the benefits associated with wetlands. Further, that despite some grey areas around the precise demarcation of use and non-use value categories, the total economic value calculation is a practical method within overall sustainability policy constrains. Taking wetland management decisions on the basis of the ecosystem function and extended cost-benefit analysis approach is both necessary and sufficient to achieve sustainability requirements such as the maintenance of “functional diversity”. Wetlands therefore continue over the long term to provide a range of valuable goods/services and are also left with a reasonable degree of resilience to counter stress and shock events, from within and outside their catchment.

CONTENTS

ECONOMIC VALUATION OF MULTI-FUNCTIONAL WETLANDS: METHODS AND TECHNIQUES

1. INTRODUCTION

Although wetlands perform many functions and are potentially very valuable, these values have often been ignored, with the result that degradation of wetlands has occurred. The debate over what is the value of wetlands, or of the environment and nature more generally, has highlighted the fact that the concept is complex and multidimensional. An economic perspective on wetlands portrays them as natural assets providing a flow of goods and services, physical as well as aesthetic, intrinsic, and moral. While it can be argued that biodiversity has intrinsic value in and of itself (either assigned by humans, or more controversially possessed regardless of human recognition) this report does not accept as a consequence that allocation decisions involving environmental assets should be decided solely by non-economic means (O'Neil), 1997; Sagoff, 2004). Deliberative processes need not be seen as substitutes for economic cost-benefit analysis. The latter can better inform the former in a complementary relationship.

The main problem when including the full range of wetland goods and services in economic choices, is that many of these goods and services are not valued on markets. There is a gap between market valuation and the economic value of many wetland functions. The non-marketed gaps must first be identified and then where possible monetised. In the case of many of the functions, the identification of economically relevant services is of special importance, since over time those services not allocated by the market have continuously gained in significance as society has evolved.

The main objective of this report is to examine and make initial recommendations on the incorporation of economic valuation methods and techniques into wetland ecosystem assessment in the UK. The wetlands assessment procedure should simultaneously be consistent with existing Treasury and other official project appraisal (DEFRA PAG 3), policy and programme assessment guidelines. The evaluation of the flood risk management options and processes is a particular focal issue; as is the link to the requirements imposed by the EU's Water Framework Directive and its water basin management approach. In the latter context there is a need to assess the cost-effectiveness of wetland creation as one of a programme of pollution abatement, flood alleviation or water storage measures.

The report is structured in the following way. First a brief summary of the main findings is presented together with some instructions on where to locate the supporting analysis in the main text. **Sections 3, 4 and 5** then present the relevant **conceptual background. Section 3** covers the ecosystem function approach which provides the framework and linkages between ecosystems, their healthy functioning and the outcomes in terms of goods and services of benefit to human society. **Section 4** sets the assessment within a range of possible policy contexts and explores the correct procedures to adopt given the prevailing circumstances, i.e., wetland conversion, wetland creation and wetland trade-offs. **Section 5** sets out the basis of socio-economic project, policy and programme appraisal and distinguishes between costs-effectiveness analysis, cost-benefit analysis and multi-criteria analysis. **Section 6 and 7** provide guidelines on the **practical application** of the ecosystem function approach, together with some selective case studies and an empirical database.

2. SUMMARY OF FINDINGS AND RECOMMENDATIONS

This report makes the following recommendations as possible changes in emphasis and/or amendments to official guidance, where reference is made as appropriate to relevant sections of text, and where reference is made to indicate whether a recommendation is a matter for policy development or for appraisal advice.

For Policy Development:

1. an ecosystem function approach be adopted as the basis for the evaluation of wetland systems (see section 3) ;

2. given the general sustainable development policy goal, taking wetland management decisions on the basis of economic cost-benefit analysis and individual function(s) alone may not be an appropriate response; this is because the component parts of a wetland system are contingent on the existence and continued proper working of the whole; the key objective should be the maintenance of "functional diversity" i.e. a sufficient spatial etc. arrangement within a catchment that allows the wetland to continue to function and provide a range of outcomes and ensures a reasonable degree of resilience to counter stress and shock events; it is this type of reasoning that should underlie a "no net loss" policy prescription (and is supported by appeals to the existence of natural values variously described as "inherent value" or "contributory value", or "indirect use value", "primary value", or "infrastructure value") (see section 3 and figure 1);

3. the cost-benefit approach (with its inherent identification of stakeholders-gainers/losers) be retained as the foundation for any assessment process and set of procedures (see section 5 and figure 4);

4. "extended" cost-benefit analysis (as set out in Treasury guidance) be deployed and adjusted as necessary to include multiple decision criteria, perhaps resulting in a multi-criteria analysis of the more complex wetland management contexts (see section 5);

For Appraisal Advice:

5. given a "no net loss" wetlands policy, it is still the case that periodically the economic cost of such a policy stance may need to be assessed in terms of, for

example, the opportunity costs of forgone development activities, or the resource and time delay costs of compensating wetland creation schemes (see section 4);

6. the set of wetland functions and related goods/services that needs to be valued will be influenced by the circumstances in which the appraisal is taking place e.g. total wetland loss, wetland degradation and partial loss of functionality, wetland creation, wetland trade-offs involving whole wetlands, or individual functions within or across wetlands (see section 4);

7. the type of economic valuation method that can be applied will depend on the type of environmental change that is the focus of attention; this will include the distinction between replaceable (non-unique) or movable wetlands versus irreplaceable (unique) or non-moveable wetlands, together with a judgement over marginal or non-marginal change impacts (see section 4);

8. it is not necessary, for assessment purposes, to construct a detailed and complex wetland classification system. A simple typology linked to a set of wetland functions which generate outcomes (goods/services) is both necessary and sufficient for the valuation of most wetlands (see sub-section 6.1);

9. the economic (monetary) valuation of a range of wetland goods and services is a practicable and meaningful exercise, while the limits to this monetary approach should also be borne in mind especially in terms of transferring economic value data across time and geographical space (see sub-section 6.4.1);

10. some dimensions of the value society may place on wetlands will remain outside of any monetary calculus and should be accounted for in other ways such as the prior imposition by government on behalf of society of quality standards, regulations and nature conservation designation legislation and practice (see sub-section 6.4.1);

11. pragmatically, the choice of approach within the wetland assessment will be reduced to the valuation of one, or a small number, of functions (with due regard to the avoidance of double counting) as a "representative" value of the whole wetland; or the estimation of a single overall composite value, via a survey method (contingent valuation or choice experiment), to serve as the proxy for total wetland value;

12. a number of economic valuation techniques can be deployed if individual or a small number of wetland functions are being valued; but that if an overall aggregate value for any given complex wetland is required then only a survey-based willingness-to-pay (or be compensated) contingent valuation method is capable of simply yielding an appropriate monetary value (see sub-section 6.4.2 and 6.4.3);

13. if either of the two pragmatic assessment procedures above are not deemed sufficient, then there is no substitute for a detailed scientific classification and process study and economic valuation exercise requiring an interdisciplinary team of experts and a significant time and resources budget;

14. a "whole landscape" catchment scale perspective should be adopted to account for the fact that similar wetland classes may provide different mixes of "valued" goods/services depending on their spatial location in a catchment; and that these locational factors should be an inherent component of any wetlands assessment (see sub-section 6.4.1);

15. it is important to recognise that some wetland goods/services are complementary and others are competitive, or even mutually exclusive and that this must be taken into account in any economic value aggregation exercise (see sub-section 6.4.1);

16. double counting of the benefits provided by functions is an ever present danger that results in an over-valuation of a given wetland, such that the guiding principle should always be to trace through the linkages between wetland function, the provision of goods/services and a change in human welfare. It is only the last link i.e. a change in human welfare that is the subject of value (gains or loss). For example, an estuarine wetland (existing or created) will perform a number of functions one of which could be nutrients/contaminants storage via sedimentation; and because of this "natural" storage function water quality is enhanced and the public recognise an amenity benefit. In this context it is <u>not</u> correct to add a value estimate for the nutrient or contaminants storage service (say the equivalent saving in conventional sewage/water treatment costs) to a value estimate of the public environmental amenity gain linked to water quality status, as this is double counting; only the amenity benefit should be accounted for in the cost-benefit valuation exercise (see sub-section 6.4.1 and 6.4.3);

17. Given the estuarine wetland context described in point 16, the costing procedure involving savings in sewage/water treatments costs would be a valid approach if the estuary concerned was subject to for example, water quality standards; it would then be legitimate to search for the most cost-effective sets of measures necessary to meet the quality standards (the benefits of which has already been decided on behalf of society) and an abatement policy including different combinations of abatement measures including wetland creation for nutrient storage could be part of the cost-effectiveness strategy exercise (see sub-section 6.4.1 and 6.4.3);

18. a typology of values based on the total economic value concept is an appropriate way to represent the multi-faceted nature of the benefits associated with wetlands; and despite some grey areas around the precise demarcation of use and non-use value categories, that such a distinction is meaningful and practical within the boundaries set by a "no net loss" condition for designated wetlands (see sub-section 6.4.1, figure 6 and Annex C);

19. there is an important distinction between economic valuation techniques which estimate welfare changes (known as "benefits") and those which estimate costs (e.g., damage costs avoided, "defensive expenditures", replacement costs etc.) as a proxy for the welfare changes; but that while the latter are a reasonable approximation and are relatively easily calculated, they are only "valid" if replacement or repair is a perfect substitute for the original wetland/function and the costs of so doing are less than the benefits derived from the wetland/function (see sub-section 6.4.2 and table 6);

20. indirect cost-based values cannot be aggregated with direct benefit value estimates for the same wetland function (see sub-section 6.4.2);

21. the proposition that it is not always necessary to initiate a new study in a project area to determine how human welfare (value) has been affected by some wetland loss or gain, should be treated with due caution; the transfer of value data (known as "benefits transfer") from one site to another, or across time, is inherently problematic and conditioned by historical, cultural and attitudinal dynamics in society (see sub-section 6.4.2);

22. while "benefits transfer" rules are still not fully agreed, the transferring of values relating to similar wetland functions across limited periods of time (ideally up to 10 years) and between roughly equivalent socio-economic and cultural contexts is a defensible procedure; the degree of robustness present in this exercise declines with the transfer of whole wetland value estimates, and/or as the timescale and socio-economic and cultural differences widen (see sub-section 6.4.2).

3. THE ECOSYSTEM FUNCTION APPROACH TO WETLANDS ASSESSMENT

The analytical framework and the methods and techniques detailed in the report are guided by the ecosystem function approach to natural resource management. This interdisciplinary approach examines the value of wetland and water resources via the linkage between water-wetland ecosystem structures and processes and the outcomes of the functioning of such systems in terms of goods and services provided to society. It is compatible with the Convention on Biological Diversity (CBD) and its ecosystem approach which has been adopted as a fundamental delivery mechanism for progress towards sustainable development

Given the generic policy goal of sustainable development, management agencies should seek to maintain the resilience of systems, in terms of the ability to cope with stress and shock. Maintenance and/or enhancement of system resilience is, as will be discussed in the following sections, linked to the ecological concept of functional diversity and the social science analogue, functional value diversity.

Maintenance of Functional Diversity

Wetlands provide a wide range of goods and services of significant value to society, such as pollution attenuation, flood alleviation, recreation and aesthetic services. We can conceive of 'valuing' wetlands as essentially valuing the characteristics of the system, and can capture these values in an economic value framework. **Figure 1** presents a framework for an ecological-economic analysis and evaluation of the functions and values of wetlands that underlies such a management strategy. At the core of the interdisciplinary analytical framework is a conceptual model, based on the concept of **functional diversity**, which links ecosystem processes and functions with outputs of goods and services, and which can then be assigned monetary economic and/or other values.

In order to assess any wetland it is necessary to compile a complete list of all the **boundary conditions** for the wetland catchment. These are the characteristic properties that describe the wetland area in its simplest and most objective terms possible. They are a combination of generic and site-specific features. A general list would include the biological, chemical and physical features that describe a wetland, such as species present, substrate properties, hydrology, size and shape (see Annex A). However in principle this list is endless and site-specific.

Wetland Structure is then defined as the biotic and abiotic webs of which characteristics are elements, such as vegetation type and soil type. By contrast **wetland processes** refer to the dynamics of transformation of matter or energy. The interactions among wetland hydrology and geomorphology, saturated soil and vegetation more or less determine the general characteristics and the significance of the processes that occur in any given wetland. These processes also enable the development and maintenance of the wetland structure which in turn is key to the continuing provision of goods and services. These ecological concepts constitute the upper part of Figure 1. The economic worth of ecosystem structure (the plants, animals, soil, air and water stocks and flows of which it is composed) is generally more easily appreciated than that of ecosystem processes. To evaluate processes for any given ecosystem, pushes scientific knowledge to its limits. A **precautionary approach** may therefore be required in any wetland management strategy.

Ecosystem functions are the result of interactions among characteristics, structure and processes. They include such actions as floodwater control, nutrient retention and food web support. The concept of ecosystem functions and ecosystem functioning is essential in linking ecology and economy (i.e. the step between wetland functioning and wetland values which is labelled wetland uses in Figure 1). Although multiple definitions of (environmental) functions exist in the literature, they have in common that they all reflect an anthropocentric perspective on ecosystem functioning, where ecosystem characteristics, structure and processes contribute to human welfare and well-being (Hueting, 1980; de Groot, 1992).

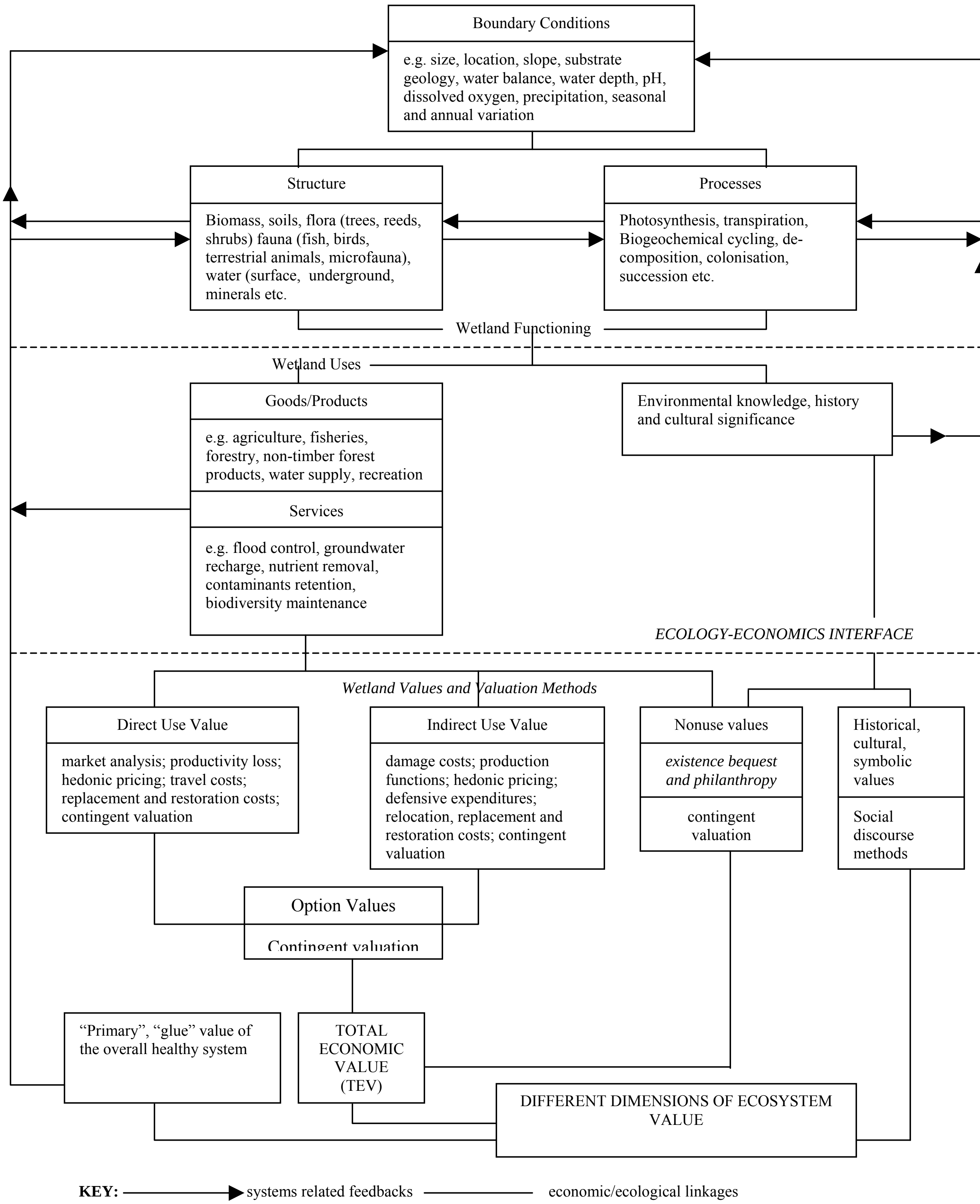


Figure 1: Wetland functions, uses and values

The conceptual model we advocate is not reductionist in the sense that it neglects the overall systems perspective that is key to the understanding of the environmental change process. Rather it is narrowly drawn in foundational terms (at the level of individual ecosystem-functions) in order to provide analytical rigour, as well as practical regulatory/policy relevance. At no time is the overall value of a healthy evolving set of environmental systems lost sight of. In this respect, a critical requirement of integrated wetland management is the introduction of planning and management mechanisms that fit at least the catchment scale (see Annex B for more information). The extent or degree to which different goods/services are deemed 'valuable' is conditioned by a diversity of **catchment-level contextual factors**. These will include human populations and access and demographic factors such as proximity, size and characteristics of human settlements; adjacent land uses within the wetland's catchment (topographic and habitat characteristics); the configuration of downstream resources; and scarcity/rarity issues at the regional scale and beyond (substitution possibilities). The human recipients of the wetland benefits will also be distributed across different spatial and temporal scales. Wetlands should therefore be seen within a catchment scale context and should be managed as part of an integrated set of resources in line with the **EC's Water Framework Directive** and its provisions.

A management strategy (e.g. coodination of CFMP's, SMP's and RBMP's) based on the principle of sustainable wetland resource utilisation should have at its core the objective of catchment ecosystem integrity maintenance, i.e., the maintenance of ecosystem components, interactions among them and the resultant behaviour or dynamic of the system. Integrity is best protected when efforts are made to secure a diverse range of wetland functions and their asset values, i.e. **functional value diversity**. The diversity of the functions provided by wetlands is dependent on the complexity and diversity of their structures and processes. These provide stability, resistance and recovery from disturbance and change. Functional diversity provides capacity for environmental-economic systems to maintain functions under stresses and shocks, building on concepts of ecosystem integrity and resilience. In this context, integrity can be defined as the maintenance of system components, the interactions between them and the resultant behaviour of the system (King, 1993).

Resilience is the system's capability to maintain stability in the presence of disturbances (often human induced), determined by the systems stability and adaptability. The maintenance of functional diversity secures a range of wetland structures and processes, which offers the best protection of the integrity of the wetland and is therefore consistent with sustainable management. From a social science perspective, a policy objective of maximum diversity maintenance also serves to ensure the maximum functional capacity and associated functional value in terms of goods and services provision. Use of the concept of functional diversity highlights the importance of the deterministic relationship between the structures and processes of a wetland and the functions that it provides. From an ecological stance, functional diversity creates variety in responses to environmental change, in particular, variety in the spatial and temporal scales over which organisms react to each other and to the environment (Steele, 1991). The onus is then on analysts and managers to take a wider perspective and examine changes in large-scale hydrological and ecological processes, together with the relevant environmental and socio-economic driving forces. Such a management strategy requires the practical coupling of economic, hydrological and ecological models.

The use of a functional approach is advocated for a number of reasons (Maltby, 1999):

- It should allow more efficient use of scarce resources by determining relationships such as the compatibility and intensity of land use activities with functioning, the capacity of ecosystems to tolerate impacts, and their resilience to human disturbance.
- Being ecosystem rather than habitat-led the approach recognises a wide range of both ecological and environmental interactions and is not restricted to a narrow view of conservation.
- Ecosystem dynamics are more easily translated into economic terms, which are usually readily understandable to the public and politicians.
- The implications of a functional approach are more appealing to the political agenda, since they extend to better use of water and land resources, improvement of environmental quality and human health and welfare.
- It allows scope for policy innovation.
- Assessment of ecosystem functioning should lead to more effective environmental protection. Two dimensions are relevant: optimising the use of limited financial resources; and identifying priority areas for protection, rehabilitation or restoration.

The functional perspective proposed in this report sets out a number of stages through which the analysis proceeds - see **Figure 2**. The initial stage requires a working classification of all relevant wetland types. This report recommends a simplified version of the hydrogeomorphic method (HGM) (see sub-section 6.1). Once this classification is established the set of functions provided by particular wetland ecosystems needs to be identified (see section 6.2).

STAGE 1	STAGE 2	STAGE 3	STAGE 4
WETLAND CLASSES →	WETLAND FUNCTIONS →	FUNCTIONING OUTCOMES →	VALUATION/ EVALUATION
		[goods/services]	[Methods/techniques]
[see section 6.1 & Tables 1 and 2]	[see section 6.2 & Table 3]	[see section 6.3 & Table 4 and 5 and Figures 1 and 5]	[see section 6.4 & Tables 6, 7 and 8]

Figure 2: Ecosystem function approach

The third stage involves the matching of functions and/or combinations of functions with outcomes in terms of tangible and intangible goods and services utilised or appreciated (i.e. welfare affecting) by human society (see section 6.3). The final stage is concerned with the valuation of the goods and services provided by the wetlands (see section 6.4). To fit in with current UK appraisal practice, economic valuations are emphasised, but this overall method can be adapted to a broader evaluation framework with multiple criteria not just economic efficiency benefits. Before considering each of these stages in more detail we first need to consider the policy appraisal context and approaches in which wetlands assessment is undertaken.

4. POLICY APPRAISAL PERSPECTIVES

In this sub-section we highlight the complications that the precise policy context in which the wetlands assessment is taking place generates and the implications for the level of detail subsequently required in the assessment process. The US HGM approach to wetlands assessment is based on **reference wetlands** i.e. fully functional examples of a wetland class/sub-class that have been relatively unaffected by human attention and therefore continue to function at a high level across a suite of identified functions. The reference wetland concept is then used to distinguish between wetland classes and within classes in terms of full or partially functioning wetland sites. The HGM approach is a very data intensive procedure and it also requires a specialist interdisciplinary team of experts to keep it operational. This level of detail is necessary because of the type of wetland assessments that are required under US legislation and in particular to help in making trade-off decisions amongst wetlands or wetland functions, often in mitigation banking/compensation cases. The UK has adopted a broad minimum standard of “no net loss” of biodiversity within the context of its international commitment to conserve biological diversity, its own nature conservation measures and relevant EU Directives.

It is therefore important, before embarking on the actual appraisal exercise, to set out a range of possible policy contexts (with an increasing data and expert knowledge cost burden) within which wetlands assessment may be required: e.g.

- contexts in which wetlands maybe destroyed and all functions are lost – this presents a relatively straightforward valuation problem in which the wetland’s conservation benefits are compared with the benefits (or forgone opportunity costs) of the development option;

- contexts in which wetlands are degraded, such that partial function loss is incurred – a more complex valuation problem involving more scientific data and economic value interdependencies; valuation is complicated by the fact that ecosystems are characterised by multiple, interdependent services that possibly exhibit complex dynamics and discontinuities around critical thresholds (Limburg *et al.* 2002; Holmes *et al.* 2004). So a decision is needed to either value a <u>set</u> of ecosystem services holistically, via a contingent valuation study for example, or whether to focus valuation on trade-offs between specific services, via attribute-based stated preference or other methods (Holmes and Adamowicz, 2003);

- Contexts in which new wetlands are created – the difficulty over valuation will depend on how simple or complex, in terms of functions provided, the new wetland is supposed to be;

- contexts in which wetlands are converted and replaced with 'functionally equivalent' created wetlands – more complex valuation problem which requires cost estimates for wetland construction/restoration and estimates of time lag costs i.e. the social benefits forgone in the time taken for a newly created wetland to gain functional equivalency with the established site;

- contexts in which existing wetlands need to be prioritised in order to allow trade-off decisions – requires full evaluation procedure;

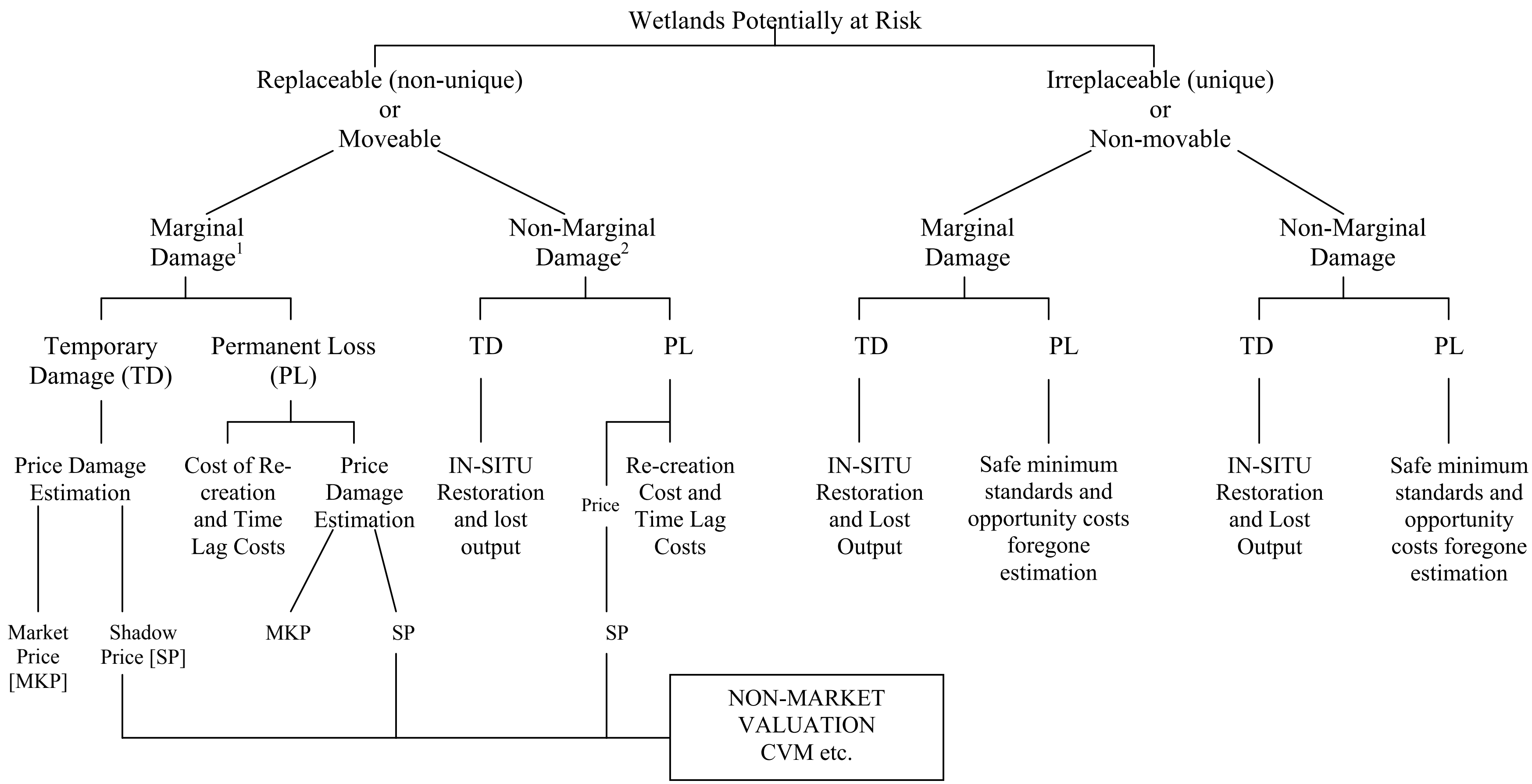
- contexts in which one or more wetland functions need to be prioritised or traded-off, both within individual wetlands or across wetlands – requires full evaluation procedure.

The type of economic valuation method that needs to be applied will depend first and foremost on the type of environmental change that is the focus of attention. Figure 3 sets out a simple schema which categorises damage impacts under two main headings i.e. damage to replaceable and irreplaceable wetlands. It is then necessary to distinguish marginal damage which affects part of a wetland or one or more functions and non-marginal damage which removes an entire wetland. Finally, the damage impact needs to be split into temporary and permanent types. The figure then shows which type of valuation method, market-based or surrogate/shadow pricing, needs to be deployed; or whether a standards and regulations approach is more appropriate. In the latter case such standards still need to be 'costed' in terms of opportunity costs foregone when wetland conservation is the a priori chosen option. We discuss the various valuation techniques and the economic valuation of particular wetland functions and corresponding outputs later.

Clearly the level of complexity is related to how the prevailing regulations and legislation are interpreted e.g. how a "no net loss" or "net gain" wetlands policy objective is actually pursued in practice. In this scoping study report it has been assumed that the wetland assessment that could be adopted would represent an initial incremental step forward from existing appraisal practice and not a full blown "expert assessment" system. With these caveats in mind we set out below a "reduced form" simple version of the HGM approach. Another complication which is related to the institutional arrangements operating at any given time or place (designations on nature conservation or other grounds, environmental regulations, legislation and custom and practice) centres around the problem of reflecting rarity/endangered habitats or cultural/symbolic significance etc. in any wetlands assessment system. For the purposes of this report it is assumed that the existing nature etc. designations process adequately reflects rarity and other characteristics of wetland sites.

Finally, from an economic efficiency perspective, it is unclear whether any decision to rely on mitigation wetland procedures if established wetlands are converted, incurs less social cost than conserving the established site. Wetland creation costs and time lag costs will tend to increase in line with the complexities and difficulties posed by the substitution of one or more wetland functions. On economic grounds, it is clear that wetland conservation should be chosen if the substitution costs exceed the foregone opportunity costs of development (Gutrich *et al.*, 2004).

Where a wetland is under pressure from human activity that provides measurable economic benefits to society, it will be necessary to illustrate the economic value of the functions performed by the wetland. The provision of such economic information is essential if an efficient level of wetland resource conservation, restoration or re-creation is to be determined. Maintaining a wetland will rarely be entirely costless. There will be costs associated with forgoing other uses of the land or with limiting activities, which might impinge upon the ability of the wetland to continue functioning. Hence the importance of making explicit the value of the multiple functions that wetlands perform, and of assessing this value within a framework which allows comparison with the gains to be made from activities that might threaten wetlands. This should serve not only to better protect these threatened ecosystems but also to improve decision making for the benefit of society. Economic valuation is therefore a logical extension to any assessment of the functions performed by wetlands for the purpose of public decision making.



[1]loss of one or more individual wetland function
[2]loss of entire wetland ecosystem

Figure 3: Wetland damages and valuation

5. EXISTING SOCIO-ECONOMIC APPRAISAL APPROACHES

The main approaches, which can form the methodological basis for strategic socio-economic option appraisal, are (see Figure 4):

- Cost effectiveness analysis
- Cost benefit analysis
- Multi-criteria analysis

Figure 4 (which should be read vertically from top to bottom) summarises the thinking behind the "cost-benefit" approach to wetlands policy appraisal. The analysis begins with an economic efficiency analysis of the gains and losses involved and a time horizon over which these costs and benefits will be incurred. An adjustment is made (discounting) to account for the assumption that future costs and benefits are valued less highly than more immediate costs and benefits. This analysis is then subjected to a sustainability filter which introduces other decisions criteria such as equity (i.e. who gains or loses in society or environmental significance, rarity etc. The former criterion (efficiency) requires that the costs and benefits are assigned weights and the latter (sustainability) that standards or regulations could be imposed to enforce conservation status and maintain ecosystem diversity. The imposition of such environmental standards should itself be subject to a cost-effectiveness exercise to determine "value for money" outcomes. Finally, in complex wetland/policy contexts a formal multi-criteria analysis (MCA) might be deployed to assess options on the basis of a number of criteria. MCAs all require some forms of weighting and scoring system applied to the different impacts that are involved in the wetland change context. The key point here is that CBA as a generic framework and economic efficient analysis, cost-effectiveness analysis and MCA need not be substitutes for each other. They are complementary and should be deployed as necessary (given precise policy circumstances) within the overall strategy set out in Figure 4.

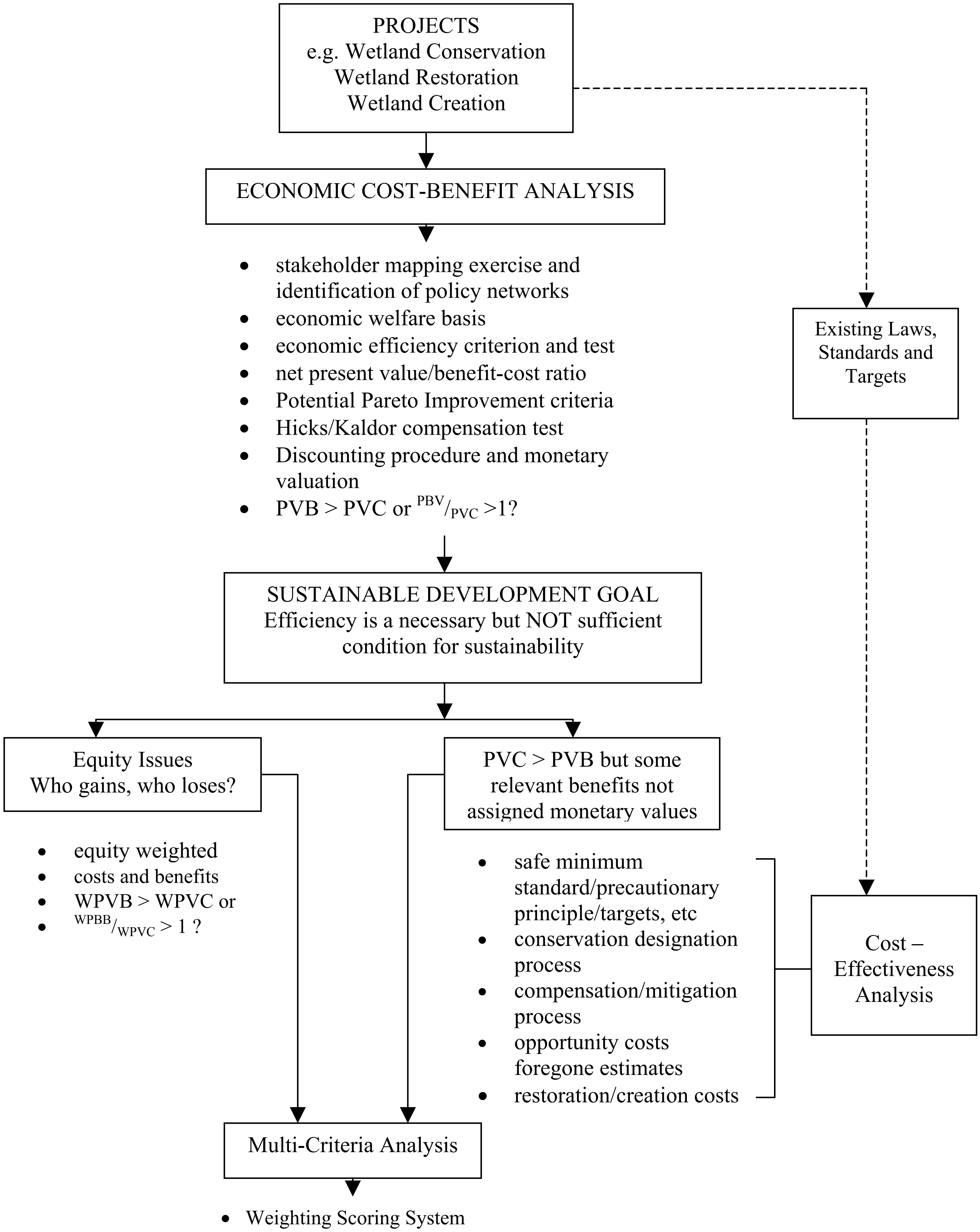


Figure 4: Project appraisal framework CBA+MCA

Usually all of the analytical approaches will involve some form of **stakeholder analysis** – in that they can involve stakeholders at a number of different points within the appraisal process. Stakeholders could for example be involved in the setting of management objectives, or in the determination of values. Deciding how stakeholders should be involved is thus a key issue, together with the identification of the linkages between stakeholders, government and other official agencies (i.e. the prevailing policy networks).

Suggested changes in management practices of the resource, arising within national and international environmental regulation, may reduce or reinforce conflicts between the various interests involved. Trying to satisfy all interest groups will often be difficult. From a policy point of view how these interests can be balanced in the best possible way is important. In order to be able to do that, insight is needed into what the various interests in the resource are, who the stakeholders are and what the distribution of the positive and negative effects of changes in management regimes will be.

The economic component of assessment consists of the identification and economic valuation of these positive and negative effects i.e. the costs and benefits, that will arise with the proposed management option and to compare them with the situation as it would be without the option. The difference is the incremental net benefit arising from the project investment. CBA is one of the evaluation tools developed by economists to determine whether a policy, project or action is economically efficient. Its principle feature is that all the pros and cons of a project, if technically possible including social and the socio-cultural and historical contexts that surround particular value gain/loss are translated into monetary terms. As a rule, a project is efficient if total benefits exceed total costs.

An important element of any assessment should be to consider the validity and reliability of economic indicators of the values people hold for environmental changes as a result of suggested management options. In those cases where a valid and reliable measure of these values (benefits) cannot be estimated, an efficient allocation of resources cannot be determined by this type of analysis. Therefore, the policy or management objective must be determined on some other basis. Once that objective is specified, the analysis tells us what the consequences are in terms of costs of choosing between different means of achieving that objective. This is called cost-effectiveness or least cost analysis.

Both cost benefit analysis and cost effectiveness analysis consider the implications of adopting an option in terms of the impacts on individuals or groups of people. They may therefore fail to take into account a range of indirect and secondary effects, which may be important in certain circumstances. Additional assessments may be required such as those provided within a multi-criteria analysis. This covers a range of techniques for assessing decision problems characterised by a large number of diverse attributes. Multi-criteria analysis (MCA) aims at providing a means for aggregating information into a single indicator of relative performance based on the full range of performance attributes. Compared to CBA, the fundamental difference lies in the recognition that economic efficiency often is not the sole objective of policy. MCA offers one way of combining expert and non-expert scientific understanding knowledge and values in order to illuminate policy trade-offs and aid decision making.

6. ECOSYSTEM FUNCTION APPROACH: ANALYTICAL STAGES

Following the schema as set out in Figure 2 earlier a number of sequential stages can be identified:

6.1 STAGE 1. Wetland Classification: The simplified hydrogeomorphic method (HGM)

Wetland classification research has a relatively long history across a number of countries, but in the past two decades scientists and government agencies in the United States have been developing a number of very detailed wetland classification systems for the functional assessment for wetlands (Cowardin *et al.*, 1979). The therefore seems the most appropriate knowledge base to work with. There are currently three assessment methods used within the US which are representative of the methods available or being used by wetland managers and planners. These are the Wetland Evaluation Technique (WET), the Environmental Monitoring Assessment Programme (EMAP-Wetlands) and the Hydrogeomorphic (HGM) approach. WET assigns specific functions of individual wetland a value, a probability rating reflecting the likelihood that a given wetland perform a function on the basis of its characteristics. The EMAP-Wetlands programme focuses on determining the ecological condition of a population of wetlands in a region. It does this by comparing a statistical sample of wetland to a reference wetland in a region. The final assessment method – the HGM approach – combines the features of the other two methods; the functions of individual wetlands are measured and are also compared to the functions performed by other wetlands. It is this final approach, while not directly applicable to wetland valuation, which could be used as a basis for the development of a practical way forward (King *et al.* (2000)). A characteristic of the HGM method is that wetland ecosystems are the unit of assessment <u>not</u> individual functions, so that overall ecosystem heath/integrity is appraised.

The HGM approach is itself in part a reaction to these earlier approaches in that it is meant to be a simpler and more practical procedure. The HGM classification is based on three fundamental factors that condition wetland functioning: the position in the landscape (geomorphic setting), the water source (hydrology) and the flow and fluctuation of the water once in the wetland (hydrodynamics). In the USA, seven major classes have been identified – see Table 1.

Table 1: HGM Wetland classification*

HGM CLASS	DOMINANT WATER SOURCE	DOMINANT HYDRODYNAMICS
RIVERINE (floodplains/riparian corridors)	OVERBANK FLOW FROM CHANNEL	UNIDIRECTIONAL, HORIZONTAL
DEPRESSIONAL	RETURN FLOW FROM GROUNDWATER AND INTERFLOW	VERTICAL
SLOPE (fens)	RETURN FLOW FROM GROUNDWATER	UNIDIRECTIONAL, HORIZONTAL
MINERAL SOIL FLATS	PRECIPITATION	VERTICAL
ORGANIC SOIL FLATS (peat bogs)	PRECIPITATION	VERTICAL
ESTUARINE FRINGE (bays/estuaries, incl. saltmarshes and intertidal areas)	OVERBANK FLOW FROM ESTUARY	BIDIRECTIONAL, HORIZONTAL
LACUSTRINE FRINGE (lakesides)	OVERBANK FLOW FROM LAKE	BIDIRECTIONAL, HORIZONTAL

* Excludes man-made wetlands
Source: King *et al.* (2000)

Adapting this classification to UK conditions and simplifying the approach further results in a “reduced form” typology – see Table 2.

Table 2: Simplified wetlands classification

WETLAND CLASSES*

SALT WATER DOMINANT	FRESHWATER DOMINANT
• ESTUARINE AND COASTAL DOMINANT (bays, estuaries and intertidal areas)	• RIVERINE (floodplains/riparian corridors, marshes)
	• DEPRESSIONAL (e.g. Norfolk and Suffolk Broads)
	• SLOPE (fens)
	• ORGANIC SOIL FLATS (peat bogs)
	• LACUSTRINE FRINGE (lakesides)

* Excluding man-made wetlands: aquaculture ponds; farm-based water bodies, settling ponds, reservoirs etc.

The next stage is to link different categories of wetlands to a set of typical functions that are routinely provided by a healthy ecosystem.

6.2 STAGE 2. Wetland Ecosystem Functioning and the Functions of Wetlands

The structures and processes of a healthy ecosystem determine a variety of typical functions that are categorised according to whether they are hydrological, biogeochemical or ecological functions.

Hydrological functions refer to the wetland’s ability to store floodwaters, the interactions between ground and surface waters and the storage of sediments:

- Flood water detention: the short and long term detention and storage of waters from overbank flooding and/or slope runoff.
- Ground water recharge: the recharge of groundwater by infiltration and percolation of detained floodwater into an aquifer.

- Ground water discharge: the upward seepage of groundwater to the wetland surface.
- Sediment retention: the net retention of sediments carried in suspension by waters inundating the wetland from river overbank flooding and runoff from a contributory area.

Biogeochemical functions of a wetland refer to the export and storage of naturally occurring chemical compounds that can have significant effects on the quality of the environment:

- Nutrient retention: the storage of excess nutrients (nitrogen and phosphorus) via biological, biochemical and geochemical processes in biomass (living and dead) and soil mineral compounds of a wetland.
- Nutrient export: the removal of excess nutrients (nitrogen and phosphorus) from a wetland via biological, biochemical, physical and land management process.
- In situ carbon retention: the retention of carbon in the form of partially decomposed organic matter or peat in the soil profile due to environmental conditions that reduce rates of decomposition.
- Trace element storage and export: the storage and removal of trace elements from a wetland via biological, biochemical and physical processes in the mineral compounds of wetland soils.

Ecological functions relate primarily to the maintenance of habitats within which organisms live:

- Ecosystem maintenance: the provision of habitat for animals and plants through the interaction of physical, chemical and biological wetland processes (including habitat and biological diversity). Nursery for plants, animals, micro-organisms.
- Food web support: the support of food webs within and outside a wetland through the production of biomass and its subsequent accumulation and export.

A review of the literature has highlighted a convenient overlap of all these functions provided by most of the different wetland classes – see Table 3.

Table 3: Wetland classes and representation functions sets

WETLAND CLASSES	TYPICAL FUNCTIONS PROVISION[1] [potentially available]
SALTWATER:	
ESTUARINE AND COASTAL FRINGE –	FLOOD CONTROL/STORM PROTECTION
	SHORELINE STABILISATION/EROSION CONTROL
	SEDIMENT, NUTRIENTS, CONTAMINANTS
	RETENSION/STORAGE → WATER QUALITY
	BIOMASS EXPORT
	CARBON SEQUESTRATION
	FISH NURSERIES AND OTHER SERVICES
	RECREATION/TOURISM
	CULTURE/HERITAGE
FRESHWATER:	
RIVERINE	WATER STORAGE/GROUNDWATER RECHARGE/ DISCHARGE
DEPRESSIONAL	FLOOD CONTROL
SLOPE	SEDIMENT ETC. RETENTION/STORAGE → WATER QUALITY
ORGANIC SOIL FLATS	RECREATION/TOURISM
LACUSTRINE FRINGE	BIODIVERSITY MAINTENANCE (incl. fisheries)
	CULTURE/HERITAGE
	CARBON SEQUESTRATION

[1] Strictly what is listed is a combination of ecological functions and related outcomes in terms of services and goods; a more precise categorisation is provided in Table 4 and 5.

The relationships between functions are both hierarchical and often complex.

e.g. element cycling		habitat provision
↓		↓
nutrient cycling		group of species
↓	or	↓
nitrogen cycling		single species
↓		
denitrification		

There may be close linkages between functions within wetland classes but also to a lesser extent between classes. For example, the detention of surface water in a wetland affects particulate retention and nutrient cycling, while also providing feeding opportunities for fish and aquatic organisms. The outcomes of their functioning over time in an integrated ecosystem can be manifested both on site and of site, as human welfare is impacted (positively or negatively) in direct or indirect ways.

6.3 STAGE 3. Wetland Functioning Outcomes

What counts at this stage of the assessment process is whether human welfare is increased or decreased due to the provision of, or changes in the provision of wetland derived goods and services. While the interrelationships between wetland functions can be complex, some of this complexity can be reduced where more aggregative impact/welfare effects are determined and then valued. It is often the case that a combination of functions within a wetland will combine to produce a service which humans value. The strength of the HGM approach is evident because it focuses the analysis on the importance of ecosystem integrity rather than individual functions. From the economic valuation perspective, complementarity or substitutability relationships between functions and the danger of double counting of the benefits provided by functions are key issues (see section 6.4).

Some of the wetland goods and services are generated within the wetland itself, **on-site**, while a much wider range are provided **off-site** see Figure 5.

ON-SITE	OFF-SITE
↓	↓
WETLAND CLASSES	WETLAND CLASSES
↓	↓
WETLAND FUNCTIONS	WETLAND FUNCTIONS
↓	↓
WETLAND OUTCOMES (goods/services)	WETLAND OUTCOMES (goods/services)
↓	↓
ECONOMIC BENEFITS (monetary values etc.)	ECONOMIC BENEFITS (monetary values etc.)
↓	↓
LOCAL-REGIONAL-GLOBAL RECIPIENTS	LOCAL-REGIONAL-GLOBAL RECIPIENTS

ACTIVE USE VALUES:	NON-USE VALUES:	ACTIVE USE VALUES:	NON-USE VALUES:
e.g. harvesting reed/sedge/hay; grazing; wildfowling; fish nurseries.	existence; bequest; option.	e.g. water storage; flood alleviation; biodiversity protection; recreation/tourism; carbon sequestration; nutrient/contaminants storage/retention → water quality	existence; bequest; option; cultural/historical/ symbolic.

Figure 5: On-site and off-site wetland goods/services

Table 4 brings together the wetland ecosystem characteristics, the functions and their outcomes in terms of goods and services and matches them to typical environmental change drivers and pressures. It offers a practical way to set out the relevant contexts and wetland functions that require assessment. It also emphasises the importance of at least a catchment wide perspective in the assessment process.

Finally Table 5 makes more explicit the link between any given ecosystem function (left hand column), or combination of functions, and the outcomes in terms of selective services and goods which impinge on human welfare. The last stage in this assessment process is the evaluation exercise itself. If an economic stance is taken then the valuation of the goods and services provided by wetlands is undertaken through the common medium of money values.

In the next section we bring together the ecological and economic dimensions in order to make explicit the links between wetland goods/services provision and their reflected value in society.

Table 4: Wetland functions and associated socio-economic benefits

Biophysical Structure or Process → Maintaining Function	Function →	Socio-Economic Benefits	Threats
	Hydrological Functions		
short and long term storage of overbank flood water and detention of surface water runoff from surrounding slopes	flood water detention	natural flood protection alternative, reduced damage to infrastructure (road network etc.), property and crops	conversion, drainage, filling and reduction of storage capacity, removal of vegetation
infiltration of flood water in wetland surface followed by percolation to aquifer	groundwater recharge	water supply, habitat maintenance	reduction of recharge rates, overpumping, pollution
upward seepage of ground water through wetland surface	ground water discharge	effluent dilution	drainage, filling
net storage of fine sediments carried in suspension by river water during overbank flooding or by surface runoff from other wetland units or contributory area	sediment retention and deposition	improved water quality downstream, soil fertility	channelization, excess reduction of sediment throughput
	Biogeochemical Functions		
uptake of nutrients by plants (N and P), storage in soil organic matter, absorption of N as ammonium, absorption of P in soil	nutrient retention	improved water quality	drainage, water abstraction, removal of vegetation, pollution, dredging
Flushing through water system and gaseous export of N	nutrient export	improved water quality, waste disposal	drainage, water abstraction, removal of vegetation, pollution, flow barriers
Organic matter accumulation	in situ retention of C	Preventing acceleration of green house effect, fuel, Paleo-environmental data source	overexploitation, drainage
Storage and removal of trace elements via biological, biochemical and physical processes in the mineral compounds of wetland soils	Trace element storage and export	improved water quality	drainage, water abstraction, removal of vegetation, pollution, dredging
	Ecological Functions		
provision of microsites for macro-invertebrates, fish, reptiles, birds, mammals and landscape structural diversity	habitat for (migratory) species (biodiversity)	fishing, wildfowl hunting, recreational amenities, tourism	overexploitation, overcrowding and congestion, wildlife disturbance, pollution, interruption of migration routes, management neglect
provision of microsites for macro-invertebrates, fish, reptiles, birds, mammals	nursery for plants, animals, micro-organisms	fishing, reed harvest	overexploitation, overcrowding and wildlife disturbance, management neglect
biomass production, biomass import and export via physical and biological processes	food web support	farming	conversion, extensive use of inputs (pollution)

Source: Modified from Turner *et al*. (1997) and Burbridge (1994).

Table 5: Wetland functions and services outcomes

Function \ Service	Potable water for residential use	Water for landscape and turf irrigation	Water for agricultural crop irrigation	Water for livestock watering	Water for food product possessing	Water for other manufacturing processes	Water for power plants	Water transport	Prevention of saline intrusion	Water/soil support for prevention of land subsidence	Natural erosion, flood and storm protection	Shoreline stabilization	Sediment removal	Transport, treatment and medium for wastes and other by-products of human activities	Improved air quality through the support of living organisms	Improved water quality through the support of living organisms	Biological diversity provision	Recreational swimming, boating, fishing hunting, trapping and plant gathering	Commercial fishing, hunting, trapping and plant gathering	Energy production	On and off site observation and study for leisure, education and scientific purposes.	Micro-climate regulation	Macro-climate regulation	Toxicant removal	Toxicant export	Cultural value provision	Historical value provision	Aesthetic value provision	Wilderness value provision
Groundwater Discharge	■	■	■	■	■	■	■	■						■	■		■	■	■		■	■							
Groundwater Recharge	■	■	■	■	■	■	■		■	■					■	■	■	■			■								
Flood Control										■	■						■				■								
Sediment Retention	■							■		■	■	■	■		■	■	■	■	■		■								
Nutrient retention														■	■	■	■	■	■		■							■	
Nutrient export														■	■	■	■	■			■								
Trace Element storage																■	■	■	■		■			■					
Trace element export																■	■				■				■				
Carbon sequestration																	■	■		■	■		■						
Biodiversity Maintenance															■	■	■	■	■	■	■	■						■	■
Culture/heritage																		■			■					■	■		

6.4 STAGE 4. Wetlands Valuation/Evaluation

The final stage of the ecosystem function approach is concerned with the monetary valuation of the goods and services provided by wetlands. As we shall see the economic valuation of a range of wetland goods and services is a practicable and meaningful exercise. Nevertheless, the limits to this monetising approach should also be borne in mind, especially in terms of transferring economic value data across time and geographical space; and the existence of intangible natural values of cultural, historical and even ethical significance.

6.4.1 Total economic value and total system value

Given the multi-faceted nature of benefits associated with wetlands there is a need for a useable typology of the associated social, economic and cultural values. In this report, we mainly focus on socio-economic values. These values depend on human preferences, i.e. what people perceive as the impact wetlands have on their welfare. In general, the economic value of an increased (or a preserved) amount of a good or service is defined as what individuals are willing to forego of some other resources in order to obtain the increase (or maintain the status quo). Economic values are thus relative in the sense that they are expressed in terms of something else that is given up (the opportunity cost), and they are associated with the type of incremental changes to the status quo that public policy decisions are often about in practice. Furthermore, who gains or losses as the result of decisions about wetlands will also be a significant element in actual decision making and from a sustainability perspective, equity issues are as important as economic efficiency issues. Before considering how to evaluate the economic value of the functions performed by a wetland in practice, it will be useful to consider how and to what extent the concept of economic value captures the variety of wetland values.

When considering environmental values, economists have generally settled for a taxonomy (see figure 6), the components of which add up to total economic value (TEV). The key distinction made is between use values and a remainder called non-use value. The latter component reflects value in addition to that which arises from usage. Thus individuals may have little or no-use for a given environmental asset or attribute but would nevertheless feel a ‘loss’ if such things were to disappear. However the boundaries of the non-use category are not clear cut and some human motivations which may underlie the position that the asset should be conserved ‘in its own right’, and labelled existence value, are arguably outside the scope of conventional economic thought. In practice, what is at issue here is whether it is meaningful to say that individuals can assign a quantified value to the environmental asset, reflecting what they consider to be intrinsic value; or even more profound whether such intrinsic value exists regardless of any human appreciation.

<table>
<tr><td rowspan="9">TOTAL ECONOMIC VALUE</td><td rowspan="4">Use value</td><td>Consumptive use value</td></tr>
<tr><td>Recreational, aesthetic and educational use value</td></tr>
<tr><td>Distant use value</td></tr>
<tr><td>Indirect use value</td></tr>
<tr><td rowspan="3">Non-use value</td><td>Existence value</td></tr>
<tr><td>Bequest value</td></tr>
<tr><td>Philanthropic value</td></tr>
<tr><td rowspan="2"></td><td>Option value</td></tr>
<tr><td>Quasi-option value</td></tr>
</table>

Figure 6: Components of the Total Economic Value of Wetlands

Economic valuation can now be combined with an ecosystem function (and related goods and service outputs) approach to wetland valuation (as shown in Figure 1). It is important to note that what is therefore being valued is not the wetland per se, but rather independent elements of ecosystem goods and services provided by wetlands. The aggregation of the main function based values provided by a given environmental ecosystem has been labelled Total Economic Value (TEV). But the aggregate TEV of a given ecosystem's functions, or combinations of such systems at the landscape level, may not be equivalent to the **total system value**. There are other sets of values that are supplementary to total economic value. First, there are values that represent the role of wetlands in natural systems. These include the value of services that stabilise natural systems and perform protective and supportive roles for economic systems. In addition, there are socio-cultural and historical dimensions of natural values that TEV may be unable to account for. Some environmental analysts also claim that natural systems have non-anthropocentric intrinsic value and that non-human species possess moral interests or rights. These environmentalist positions lead to the advocacy of environmental sustainability standards or constraints, which to some extent obviate the need for valuation of specific components of the environment. It is still necessary, however, to quantify the opportunity costs of such standards, or to quantify the costs of current, and prospective environmental protection and maintenance measures – see figure 4. A detailed discussion of the variety of natural values associated with wetland ecosystems and the limitations of the TEV concept is found in Annex C.

The ability to value wetland ecosystem goods and services is also constrained by the complexity of the wetland ecosystem itself. In particular it is necessary to take into account the following issues:

- The spatial and temporal scale. Ecosystem processes operate over a range of spatial and temporal scales; the scales that are appropriate to study the management of one process may not be suitable for other processes.

- The persistence of the functions of a wetland are dependent on the complexity and diversity of its structures and processes. The diversity provides resistance and recovery from disturbances and capacity for long term adaptation, as well as being a sensitive indicator of environmental change.
- Ecosystems are dynamic in space and time. Change is the normal course of events. Natural or human induced disturbance create an inter-related mosaic of change. This influences ecosystem processes at large spatial scales.
- Uncertainty and surprise are inevitable. There is much that is not understood about wetland ecosystems. Progress will yield some new knowledge, but the complexity and interactions of non-linear processes means that certain elements of wetlands will always be difficult to predict and that surprise outcomes are inevitable, hence the importance of the precautionary approach.

The "production function" of wetland ecosystems is so complex, and little understood in many instances, that reliable estimates of all services cannot be made. Human intervention in these complex and large scale systems can have results that we do not understand fully at present (Turner, 2000). An aspect of this complexity is that joint products are inherent in most wetland ecosystem processes. From an economic valuation standpoint it is thus important to recognise that some wetland goods/services are complementary, others are competitive or even mutually exclusive e.g. increased access and recreation versus conservation of breeding sites or ranges for rare/endangered species. Accounting for value must recognise all these joint product values and avoid double counting.

Finally it seems clear that all wetlands to a greater or lesser extent provide some valued goods/services but that different wetland classes may yield different mixes of goods/services. We have shown earlier that nevertheless a representative set of functions can be distinguished and fitted to a very simple wetlands classification. But it is still the case that similar wetland classes may provide different mixes of 'valued' goods/services depending on their location in different catchment contexts. These locational factors are particularly important and also serve to emphasise the importance of assessing wetlands within their relevant catchments or river basins as discussed in section 3. Relevant catchment-level contextual factors that require consideration in appraisal will include:

- Proximity of the wetland, and of the result of the functions it performs, to human settlement
- Accessibility of the wetland to humans
- Land use in the vicinity of and downstream from the wetland.
- Predominant local industries such as farming, manufacturing, mining, tourism, forestry - not limited to the catchment, but the local region
- Size and distinctiveness of the wetland in comparison with other nearby wetlands
- Likely changes in human factors in the catchment in the foreseeable future, such as urban expansion, change in farming practices, increasing nature conservation, road building, river management etc.
- The existence of nature conservation schemes within or nearby the wetland

- Effluent discharge or nutrient seepage into a river upstream or downstream of the wetland
- Known or historical problems with the water environment, such as pollution of the river, flooding episodes

In conclusion, a typology of values based on the total economic value concept is an appropriate way to represent the multi-faceted nature of the benefits associated with wetlands; and despite some grey areas around the precise demarcation of use and non-use value categories, such a distinction is meaningful and practical.

6.4.2 Economic Valuation Techniques

A range of valuation techniques exists for assessing the economic value of the functions performed by wetlands, and these are detailed in Table 6. Many wetland functions result in goods and services, which are not traded in markets and therefore remain un-priced. It is then necessary to assess the relative economic worth of these goods or services using the non-market valuation techniques shown in Table 6. More detailed information on the underlying theory of some of these techniques is given in Annex D.

An important distinction to make is between those valuation techniques, which estimate benefits directly and those which, estimate costs as a proxy for benefits. For instance, estimating Damage Costs Avoided, Defensive Expenditures, Replacement/Substitute Costs or Restoration Costs as part of an economic valuation exercise suggests that the costs are a reasonable approximation of the benefits that society attributes to the resources in question. The underlying assumption is that the benefits are at least as great as the costs involved in repairing, avoiding or compensating for damage. These techniques are widely applied due to the relative ease of estimation and availability of data, but it is important to be aware of the limitations in terms of the information they convey with respect to economic benefits. Where it can be shown that, a) replacement or repair will provide a perfect substitute for the original function, and, b) the costs of doing so are less than the benefits derived from this function, then the costs do indeed represent the economic value associated with that function.

Where market prices exist for resources, these may have to be adjusted to provide social or shadow prices as explained above, but otherwise they are likely to provide a relatively simple means of assessing economic value. Approaches related to market analysis include the assessment of productivity losses that can be attributed to changes in the ecosystem and the incorporation of the ecosystem as just one input into the production function of other goods and services. Investment by public (especially government) bodies in conserving ecosystems may represent a surrogate for aggregated individual willingness to pay and hence social value. These ‘public prices’ paid for resources have been used to approximate the value society places upon them, as for instance the costs of designating an ecosystem as a nature reserve. For a variety of reasons, these are unlikely to accurately reflect aggregated individual values, although techniques exist for attributing economic value based on such ‘collective choice’ decisions.

In the absence of market prices, two theoretically valid benefit estimation techniques would be hedonic pricing or the travel cost method. However, these are based on preferences being 'revealed' through observable behaviour, and are restricted in their application to where a functioning market exists, such as that for property, in the case of hedonic pricing, or where travel to the site is a prerequisite to deriving benefit, such as with recreational visits, in the travel cost method. Contingent valuation, based on surveys that elicit 'stated preferences', has the potential to value benefits in all situations, including non-use benefits that are not associated with any observable behaviour. The legitimacy of contingent valuation methods and results is still contested, especially in the context of non use values, and conducting a contingent valuation survey can sometimes be a lengthy and resource-intensive exercise.

It is not always necessary to initiate a new study in a project area to determine how the wellbeing of individuals might be affected by some environmental change. If a similar project has previously been undertaken elsewhere, estimates of its economic consequences might be usable as an indicator of the impacts of the new project.

Such an approach has been termed 'benefits transfer' because the estimates of economic benefits are 'transferred' from a site where a study has already been completed to a site of policy interest. The benefits transferred from the study site could have been measured using any of the direct or indirect valuation techniques outlined above. There are three broad approaches to benefits transfer to consider here. **Transferring average benefit estimates**. Here we assume that the change in wellbeing experienced on average by individuals at existing sites is equal to that which will be experienced at the new site. Previous studies are used to estimate the consumer surplus or average WTP of individuals engaged in, say, recreational activities of various kinds. The value of a 'person-day' for each recreational activity at existing sites is multiplied by the forecast change in the number of days at the new site, to obtain estimates of the aggregate economic benefits of recreation at the new site. **Transferring adjusted average benefit values**. Here the mean unit values of the existing studies are adjusted before transferral to the new site, in order to account for any biases that are thought to exist, or to reflect better the conditions at the new site. These differences might be in socio-economic characteristics of households, in the environmental change being looked at, and in the availability of substitute goods and services.

Transferring benefit functions. Instead of transferring adjusted or unadjusted average values, the entire demand function estimated at existing sites could be transferred to the new site. More information is passed over in this way.

Benefits transfer is still in its infancy, in part because for many environmental policy issues only a limited number of high quality valuation studies have been completed. However, it is potentially a very important and useful estimation approach, as it could feasibly provide accurate and robust benefit estimates at a fraction of the cost of a full-blown valuation study. Further details regarding Benefits Transfer and the problems of transferring values are given in Annex D.

Table 6: Valuation Methodologies Relating to Wetland Functions

Valuation Method	Description	Direct Use Values	Indirect Use Values[1]	Non-use Values
Market Analysis	Where market prices of outputs (and inputs) are available. Marginal productivity net of human effort/cost. Could approximate with market price of close substitute. Requires shadow pricing.	√	√	
(Productivity Losses)	Change in net return from marketed goods: a form of (dose-response) market analysis.	√	√	
(Production Functions)	Water treated as one input into the production of other goods: based on resource linkages and market analysis.		√	
(Public Pricing)	Public investment, for instance via land purchase or monetary incentives, as a surrogate for market transactions.	√	√	√[2]
Hedonic Price Method (HPM)	Derive an implicit price for an environmental good from analysis of goods for which markets exist and which incorporate particular environmental characteristics.	√	√	
Travel Cost Method (TCM)	Costs incurred in reaching a recreation site as a proxy for the value of recreation. Expenses differ between sites (or for the same site over time) with different environmental attributes.	√	√	
Contingent Valuation (CVM)	Construction of a hypothetical market by direct surveying of a sample of individuals and aggregation to encompass the relevant population. Problems of potential biases.	√	√	√
Damage Costs Avoided	The costs that would be incurred if the catchment function were not present; e.g. flood prevention.		√	
Defensive Expenditures	Costs incurred in mitigating the effects of reduced environmental quality. Represents a minimum value for the environmental function.		√	
(Relocation Costs)	Expenditures involved in relocation of affected agents or facilities: a particular form of defensive expenditure.		√	
Replacement / Substitute Costs	Potential expenditures incurred in replacing the function that is lost; for instance by the use of substitute facilities or 'shadow projects'.	√	√	√[3]
Restoration Costs	Costs of returning the degraded catchment function to its original state. A total value approach; important ecological, temporal and cultural dimensions	√	√	√[3]

Notes to table 6:

[1] Indirect use values associated with functions performed by an ecosystem will generally be associated with benefits derived off-site. Thus, methodologies such hedonic pricing and travel cost analysis, which necessarily involve direct contact with a feature of the environment, can be used to assess the value of indirect benefits downstream from the ecosystem.

[2] Investment by public bodies in conserving ecosystems (most often for maintaining biodiversity) can be interpreted as the total value attributed to the ecosystem by society. This could therefore encapsulate potential non-use values, although such a valuation technique is an extremely rough approximation of the theoretically correct economic measure of social value, which is the sum of individual willingness to pay.

[3] Perfect restoration of the ecosystem or creation of a perfectly substitutable 'shadow project' ecosystem, which maintains key features of the original, might have the potential to provide the same non-use benefits as the original. However, cultural and historical aspects as well as a desire for 'authenticity' may limit the extent to which non-use values can be 'transferred' in this manner to newer versions of the original. This is in addition to spatial and temporal complexities involved in the physical location of the new catchment or the time frame for restoration.

6.4.3 Economic Valuation of Wetland Ecosystem Functions in Practice

As discussed in the earlier sections, wetlands provide a variety of ecosystem functions that are beneficial to society. A distinction was made between hydrological, biogeochemical and ecological functions.

In evaluating the economic value of the functions performed by a wetland, a common approach is undertaken. The approach involves the identification of all the goods and services produced by each of the wetland functions, followed by the valuation of each of the goods and services.

The socio-economic benefits derived from these functions can be measured in a variety of ways, as discussed in the previous section. In doing so, the following steps can be distinguished:

1) Choice of the appropriate assessment approach (impact/damage analysis, partial valuation, total valuation).
2) Definition and specification of the spatial system boundary of the wetland.
3) Identification of the functions of the wetland ecosystem and potential goods and services provided by them.
4) Assessment of their actual provision level (including quality), and what effect there would be if the wetland were removed.
5) Identification of the groups of people in society who benefit from them (or will be suffering a loss when they are removed, destroyed or degraded).
6) Identification of the possible values attributed to them by these different groups in society.
7) Selection of the appropriate economic valuation technique(s).
8) Estimation of the total economic value.

Stages 5-8 are important because it is not the degree to which the function is being performed that apportions economic value, but the influence that the good or service will have on resources regarded by society as worth conserving. The choice of valuation technique that can be employed to estimate the economic value of the good or service (step 7) will often be up to the analyst to decide. While some methods are theoretically preferable to others (see previous section), other 'second best' measures may often be easier to determine in practice. The choice of method is likely to depend, in part, on time, resources and data available for the investigation.

In considering the economic valuation of wetland functions in practice, the following important issues also need to be born in mind by appraisers (see Annex E for a more detailed discussion):

- **The spatial and temporal scale** – it is important to determine initially what the scale of assessment is going to be since the scales that are appropriate to study the management of one process may not be suitable for other processes.
- **Aggregation and Double counting** – if each output provided by a wetland is identified separately, and then attributed to underlying functions, there is a likelihood that benefits will be double counted when aggregation is undertaken.
- **Allocation over time** – it is frequently necessary to consider wetland functions that have different temporal patterns of benefits (and costs), or that differ in their duration. Discounting provides a common matrix that enables comparison of benefits (and costs) that occur at different points in time.
- **Risk and uncertainty** – risk and uncertainty will be associated with both the physical outcomes associated with wetland ecosystem changes and their economic consequences. Assessing the possible outcomes and likelihood of perturbations to highly complex wetland ecosystems will inevitably be fraught with difficulty. The use of future **scenario analysis** is one way of mitigating uncertainty-related problems, see case study (C).
- **Irreversible change** - irreversible impacts are not accounted for in the standard procedures for economic evaluation. Under such circumstances, account needs to be taken of the uncertain future losses that might be associated with potential irreversible change. Some protection to the interests of future generations can be offered through the imposition of the safe minimum standards decision rule
- **Data limitations** - it is inevitable that some of the data required for an economic evaluation will not be readily available. Where data are limited, this should be acknowledged and the measures taken in response to this limitation clearly specified. The results and recommendations should be made explicitly conditional on these limitations.

Finally it should be noted that although a number of economic valuation techniques can be deployed to value individual or a small number of wetland functions, if an overall aggregate value for any given wetland is required then only a survey based willingness to pay contingent valuation method is capable of simply yielding an appropriate monetary value.

The application of the valuation techniques to many of the functions has already been extensively considered elsewhere (Gibbons, 1986; Young, 1996; National Research Council, 1997; Renzetti, 2002), and hence discussion of these is limited to some of the more important and less well understood of these. A summary of these functions and the techniques that are applied to them is shown in Table 7. In addition, Table 8 summarises this information according to the main wetland service outcomes.

Table 7: Valuation methods used to estimate the socio-economic benefits of wetland ecosystem functions

Wetland ecosystem function (Annex giving further details)	**Direct Use Values**	**Indirect Use Values**	**Non-use Values**	**Economic valuation technique**[1]							
				MA	**HP**	**TC**	**CV**	**DE**	**RC**	**PF**	**ADC**
Hydrological functions (F1)											
Flood water detention (F1.1)	√	√			√		√	√	√		√
Ground water recharge/discharge (F1.2/3)	√	√	√	√	√	√	√		√	√	
Sediment retention (F1.4)	√	√									√
Biogeochemical functions (F2)											
Nutrient retention/export (F2.1/2)	√	√	√	√	√	√	√	√	√	√	
Trace element storage/export (F2.3/4)	√	√	√	√	√	√	√	√	√	√	
In situ carbon retention (F2.5)	√	√	√								√
Ecological functions (F3)											
Ecosystem maintenance (F3.1)	√	√	√	√	√	√	√		√		
Food web support (F3.2)	√	√	√	√					√		

[1]Notes:
MA : Market Analysis
HP : Hedonic Pricing
TC : Travel Cost
CV : Contingent Valuation/ranking
DE : Defensive Expenditures
RC : Replacement Costs (Shadow Project)
PF: Production Function
ADC: Avoided Damage Costs

The indirect use benefits from flood water detention mainly refer to potential damage avoided downstream. Besides the calculation of avoided damage costs, defensive expenditures related to flood warning or avertive action, the replacement or substitution costs of building dikes if the wetland is removed, hedonic pricing studies have looked at the influence of risks of flooding on property prices (see Annex H for an overview of studies). Contingent valuation studies have been used to estimate public WTP for flood protection (again see Annex H for these and other studies related to the functions also discussed below).

Ground (and surface) water recharge and discharge may include non-use values when water supplies are maintained for the sake of future generations. Based on market analysis, production function approaches and the calculation of replacement costs, the direct and indirect use value of wetlands for drinking water purposes and irrigation in agriculture have been estimated. Hedonic pricing studies have investigated the effect of access to water on property prices. Travel cost and contingent valuation studies have examined the recreational benefits associated with instream flow levels.

The direct and indirect benefits of wetlands retaining sediments from flood water and runoff water have been mainly estimated in terms of damage avoided (e.g. recreation, navigation, agriculture).

The direct and indirect use benefits from nutrient and trace element retention and export mainly refer to maintaining or improving water quality for various reasons, including drinking water and recreation. Market analysis and production function approaches have been used to estimate the value of clean water on human health and commercial fishing (forgone earnings). Defensive or avertive expenditures incurred by households have been assessed to avoid exposure to contaminated (ground) water supply. Replacement cost studies have looked at the costs of equivalent waste water treatment methods. Travel cost studies have been used to assess the value of improved water quality at recreational sites, while hedonic pricing studies investigated the impact of differing water qualities on residential housing prices. Contingent valuation studies have assessed WTP for protecting and improving (ground) water quality for a variety of reasons (drinking water, health, recreation, habitat preservation etc.), including non-use motivations such as bequest or existence values.

The socio-economic value of a wetland's capacity to store carbon (peat accumulation) has been estimated through exercises looking at the damages caused by climate change. Although such studies have only assessed the indirect use value involved, the storage of carbon may also have non-use value (e.g. reducing climate change for the sake of future generations).

The socio-economic benefits of wetlands producing biomass (fish, reed etc.) have been evaluated through market analyses and replacement cost estimations of shadow projects. Restoration cost studies have looked at the costs related to rehabilitating the ecology of wetlands (reed beds, cleaning up pollution etc.). Hedonic pricing studies have been carried out to assess property prices in and near wetlands. Travel cost studies have estimated the recreational expenditures for waterfowl hunting, birdwatching, fishing etc. Finally, contingent valuation surveys have been used to estimate the use and non-use values, separately or at the same time, through public WTP for species and habitat preservation.

Annex F provides a more detailed discussion of how each of the functions is assessed in practice using the various valuation techniques. For any given assessment the type and number of functions that need to be valued will vary with the policy context in which the appraisal is being undertaken and with the choice of valuation approach.

Table 8: Wetland Service Outcomes and Valuation Techniques Used

Service Provided	Effect on economic value	Valuation techniques used to value
Potable water for residential use	Change in welfare from change in availability of potable water. Change in human health or health risks.	MA; PF; RC; ADC; CV; DE; HP
Water for landscape and turf irrigation	Change in cost of maintaining public or private property.	MA; PF; RC; ADC; CV
Water for agricultural crop irrigation	Change in value of crops or production costs. Change in human health or health risks.	MA; PF; RC; ADC; CV; DE; HP
Water for livestock watering	Change in value of livestock products or production costs. Change in human health or health risks.	MA; PF; RC; ADC; CV; DE; HP
Water for food product processing	Change in value of food products or production costs. Change in human health or health risks.	MA; PF; RC; ADC; CV; DE; HP
Water for other manufacturing processes	Change in value of manufactured goods or production costs.	MA; PF; RC; ADC; CV
Water for power plants	Change in cost of electricity generation.	MA; PF; RC; ADC; CV
Water transport	Change in economic output.	MA; PF; RC; ADC;
Prevention of saline intrusion	Change in cost of maintaining public or private property	MA; PF; RC; ADC; CV
Water/soil support for prevention of land subsidence	Change in cost of maintaining public or private property	MA; PF; RC; ADC; CV
Natural erosion, flood and storm protection	Change in cost of maintaining public or private property	MA; PF; RC; ADC; CV
Shoreline stabilization	Change in cost of maintaining public or private property	MA; PF; RC; ADC; CV
Sediment removal	Change in human health or health risks. Change in animal health or health risks. Change in economic output or production costs.	MA; PF; RC; ADC; CV; DE; HP
Transport, treatment and medium for wastes and other by-products of human activities	Change in human health or health risks. Change in animal health or health risks. Change in economic output or production costs.	MA; PF; RC; ADC; CV; DE; HP
Improved air quality through the support of living organisms	Change in human health or health risks. Change in animal health or health risks.	MA; PF; RC; ADC; CV; DE; HP

Improved water quality through the support of living organisms	Change in human health or health risks. Change in animal health or health risks. Change in economic output or production costs.	MA; PF; RC; ADC; CV; DE; HP
Biological diversity provision	Change in quantity or quality of recreational activities. Change in human health or health risks.	MA; PF; RC; ADC; CV; TC; DE; HP
Recreational swimming, boating, fishing hunting, trapping and plant gathering	Change in quantity or quality of recreational activities. Change in human health or health risks.	MA; PF; RC; ADC; CV; TC; DE; HP
Commercial fishing, hunting, trapping and plant gathering	Change in value of commercial harvest or costs. Change in human health or health risks.	MA; PF; RC; ADC; CV
Energy production	Change in economic output	MA; PF; RC; ADC;
On and off site observation and study for leisure, education and scientific purposes.	Change in quantity or quality of on/off site observation or study activities	MA; PF; RC; ADC; CV; TC
Micro-climate regulation, Macro-climate regulation	Change in human health or health risks. Change in animal health or health risks. Change in economic output or production costs.	MA; PF; RC; ADC; CV; DE; HP
Toxicant removal, Toxicant export	Change in human health or health risks. Change in animal health or health risks.	MA; PF; RC; ADC; CV; DE; HP
Cultural value provision	Change in personal utility or well-being	CV
Historical value provision	Change in personal utility or well-being	CV
Aesthetic value provision	Change in personal utility or well-being	CV
Wilderness value provision	Change in personal utility or well-being	CV

Key: MA = Market Analysis;
PF = Production Function;
RC = Replacement Costs;
ADC = Avoided Damage Cost;
CV = Contingent Valuation/Contingent Ranking;
TC = Travel Cost;
DE = Defensive Expenditures;
HP = Hedonic Pricing;

7. CASE STUDIES AND OVERVIEW OF EMPIRICAL STUDIES

Following on from the above discussion, a number of actual case studies have been outlined in detail in Annex G in order illustrate some of the procedures and challenges involved in assessing the value of wetland ecosystems and their functions. The case studies address a number of different problems across different temporal and spatial scales – from individual wetland function valuation, through whole area management in the context of a single/few environmental pressures, to catchment/landscape scale problems.

The first case study (A) consists of two examples concerned with the valuation of nature conservation, recreation and other economic interests at risk from flooding. The first example has been chosen to illustrate the use of a fully fledged contingent valuation survey to obtain monetary willingness to pay values, which assess the benefits of creserving the existing landscape, ecology and recreational characteristics of the Norfolk Broads area relative to their expected values in the absence of a Broadland wide flood alleviation scheme. The study was conducted in answer to a real-world question regarding the funding of flood defences in Broadland. The second example extends the analysis to a wider consideration of the economic costs and benefits of protection to agriculture, property and infrastructure damage-avoidance benefits of such defences. Although the benefit-cost ratio of the latter items was calculated at 0.98, when conservative measures of WTP for the recreational and environmental benefits of flood prevention are considered the benefit-cost ratio increases substantially to 1.94, indicating that the benefits of a flood alleviation strategy are almost twice the associated costs. The case study illustrates the importance that non-market values can have in providing justification for central government funding support for a proposed flood alleviation strategy.

The second case study (B) differs from the first in that it illustrates the valuation of ecosystem functions in the context of a wetland that society has already decided to conserve. It looks at how economic appraisal can be used in such circumstances to consider the cost-effectiveness of different fen management options so as to achieve protection at minimum cost. Existing practices for reedbed management are typically labour intensive and therefore costly. They may only be suitable under certain wetland conditions and can sometimes be ecologically damaging. They also tend to generate a considerable amount of waste cut material that remains on site, resulting in further deterioration of reedbed and fens. The study examines the success of an alternative wetland reed/sedge harvesting scheme, using a new prototype mechanical harvester for harvesting and transporting cut material off-site. The analysis aims to explain the variability of costs for the different options according to site characteristics, in order to predict which option is more effective at various types of sites. Although the available data are inadequate to allow a complete economic analysis, the study shows how the gross margins (financial returns) for the harvester scheme vary according to the location of the reed beds. This study serves to show the sensitivity of the financial returns from commercial exploitation to locational factors. Nevertheless, fenland is of special conservation significance and has to be managed to retain its ecological value. The assessment of such sites is therefore focussed on least cost methods of management and not on any overall positive economic cost-benefit ratio or net present values.

The third case study (C) illustrates two examples concerned with the use of economic valuation and appraisal in the context of wetland creation, one set in an inland location and one at the coast. In the latter case managed realignment is now becoming an important element in coastal zone policy, and is being increasingly utilised for flood defence purposes. The inland example at Nottingly has been selected because it includes within its appraisal an estimation of the economic value of the environmental gains consequent upon the creation of a wetland. Unfortunately, the value estimation procedure used is incorrect as it aggregates both a proxy cost estimate and a direct value estimate in order to value the environmental improvement. Technically one or other of three estimates should have been used. The result is an over-estimation of the environmental gain from the created wetland. In this particular instance, the overall cost-benefit result is not significantly affected but this may not always be the case. Fuller details of this procedural error are set out in the annex.

As a counterbalance, the coastal wetland creation example in the Humber Estuary has been included in order to show how the correct economic method should be applied and how it can be combined with futures scenarios analysis in order to provide policymakers with an array of possible options.

The final case study (D) relates to the valuation of benefits from wetland restoration. It illustrates an example of the contingent valuation method being used to assess the economic benefits of increasing the landscape, ecology and recreational characteristics of the Culm Grasslands in south west England – a scientifically important habitat supporting many endangered species including the Marsh Fritillary Butterfly and Curlew.

This study is important because it encompasses a conservation policy context in which experts value the sites and their amalgamation into a more coherent whole very highly. However, the public, were by and large ignorant of the sites and their ecological significance. The analysis illustrates how a survey based economic analysis can be deployed and can yield an aggregate/composite value estimate for the wetland as a whole; and how information can be provided to help survey respondents articulate their preferences.

Finally, Annex H presents an overview of empirical studies related to wetland function valuation. The overview, is wide ranging but not comprehensive, and does indicate that valuation techniques have been extensively applied to a large range of wetland ecosystem functions. It also indicates that although the economic values are significant, there is nevertheless wide variation across wetland valuation studies. The wide variation found in studies will mean that 'benefits transfer' is, fraught with difficulties and subject to a number of caveats, where any results and recommendations that transpire should explicitly be made conditional on these limitations. If a benefits transfer approach is chosen as part of an economic valuation then the main challenge faced is in finding the most appropriate studies to use in the transfer exercise. Criteria for transferring benefits between sites are suggested by Boyle and Bergstrom (1992) as:

- it should be the same goods or services that are being valued;
- relevant populations need to be very similar;
- the assignment of property rights concerning the wetland function under consideration should be the same.

Meta-analysis enables researchers to identify criteria for valid environmental value transfer or to test the convergent validity of value estimates (for further details on meta-analysis see Brouwer *et al.*, 1999). As more information about factors influencing environmental valuation outcomes becomes available, for instance through meta-analysis, transfers across populations and sites seems to become more practicable, using either existing (secondary) information only or supplementing this information with new original (primary) data. However, very little published evidence exists of studies that test the validity of environmental value transfer. Moreover, in the few studies that have been carried out, the transfer errors are substantial (Brouwer, 1999). Further details on the problems and recommendations for undertaking Benefits Transfer are found in Annex D.

8. CONCLUSIONS

The economic (monetary) valuation of a range of wetland goods/services is a practicable and meaningful exercise. There are, however, limits to this set of procedures and particular care needs to be taken to avoid double counting errors (given ecosystem complexities) and whenever value data transfers (over time and space) are being considered. A typology of values based on the total economic value concept is an appropriate way to represent the long-run and multi-faceted characteristics of the benefits associated with wetlands. Further, despite some ambiguities around the precise demarcation of use and non-use value categories, the total economic value calculation is a practical and policy relevant method. Taking wetland management decisions on the basis of the ecosystem function and extended cost-benefit analysis approach is both necessary and sufficient to meet the requirements of a sustainable development policy strategy.

REFERENCES

Abdalla, C. (1994) Groundwater values from avoidance cost studies implications for policy and future research. *American Journal of Agricultural Economics*. 76: 1062-1067.

Abdalla, C.W., Roach, B.A., Epp, D.J. (1992) Valuing environmental-quality changes using averting expenditures – an application to groundwater contamination. *Land Economics,* 68 (2) 163-169.

Aitchson, J. and M. Ashby (2000) The common lands of Devon: a biological survey, report – Rural Surveys Research Unit, Department for the Environment, Transport and the regions. http://www.defra.gov.uk/wildlifecountryside/issues/ Common/biosurvey/countyreports/devon.pdf

Andrews, J.E., Samways, G., Dennis, P.F. and Maher, B.A., (2000). Origin, abundance and storage of organic carbon and sulphur in the Holocene Humber Estuary: emphasising human impact on storage changes. In: *Holocene Land-Ocean Interaction and Environmental Change around the North Sea*, Shennan, I. and Andrews, J., (eds.), Geological Society, London, Special Publications, 166: 145-170.

Arrow, K., Solow, R., Schuman, H., Ragner, R. and Portney, P. (1994) Report to the NOAA Panel on Contingent Valuation. *US Federal Register*, 58 (10) 4602-4614.

Barbier E.B., Markandy A.A., Pearce, D.W. (1990) Environmental sustainability and cost-benefit-analysis. *Environmental Planning A*. 22(9) 1259-1266.

Barbier, E.B. (1994) Valuing environmental functions: tropical wetlands. *Land Economics*, 70 (2): 155-173.

Bateman I., Willis, K., Garrod G. (1994) Consistency Between Contingent valuation estimates – a comparison of 2 studies of UK National Parks. *Regional Studies* 28 (5) 457-474.

Bateman, I.J. and Langford, I. H. (1997) Non-users' willingness to pay for a national park: An application and critique of the contingent valuation method, *Regional Studies*, 31(6): 571-582

Bateman, I.J., I.H. Langford, R.K. Turner, K.G. Willis, and G.D. Garrod. (1995) Elicitation and truncation effects in Contingent Valuation Studies. *Ecological Economics*, 2, 161-179.

Bateman, I.J., Langford, I.H. and Rasbash, J. (1999) Elicitation effects in contingent valuation studies, in Bateman, I.J. and Willis, K.G. (eds.) *Valuing Environmental*

Preferences: Theory and Practice of the Contingent Valuation Method in the US, EU, and Developing Countries, Oxford University Press, p511-539.

Bateman, I.J., Langford, I.H., Willis, K.G., Turner, R.K. and Garrod, G.D. (1993) The impacts of changing willingness to pay question format in contingent valuation studies: An analysis of open-ended, iterative bidding and dichotomous choice formats. GEC working paper 93-05, Centre for Social and Economic Research on the Global Environment (CSERGE), University of East Anglia and University College London.

Bateman, I.J., Willis, K.G., Garrod, G.D., Doktor, P., Langford, I.H., and Turner, R.K. (1992) Recreation and environmental preservation value of the Norfolk Broads: A contingent valuation study. Report to the National Rivers Authority, Environmental Appraisal Group, University of East Anglia, pp. 403.

Begon, M., Harper, J.L., and Townsend, C.R., (1996) Ecology. Blackwell Science: Oxford.

Bishop, R.C. (1978) Endangered species and uncertainty: The economics of a safe minimum standard. *American Journal of Agricultural Economics*, 60: 10-18.

Boadway, R.W. and Bruce, N. (1984) Welfare Economics. Basil Blackwell, Oxford.

Boddington M.A.B. (1993) Financial and economic measurement of environmental factors. *Journal of the Institute of Water and Environmental Management*, 7: 125-133

Boyle, K.J. and Bergstrom, J.C. (1992) Benefits transfer studies: Myths, pragmatism, and idealism. *Water Resources Research*, 28(3): 657-663.

Broads Authority (1997) The strategy and management plan for the Norfolk and Suffolk Broads, February 1997.

Brookshire, D.S. d'Arge, R., Schulze, W. and Thayer, M. (1982). Valuing Public Goods: A Comparison of the Survey and Hedonic Approaches. *American Economic Review*, 72(1): 478-88

Brouwer (1998) Future Research Priorities for Valid and Reliable Environmental Value Transfer. Global Environmental Change Working Paper 98-28, Centre for Social and Economic Research on the Global Environment (CSERGE), University of East Anglia and University College London.

Brouwer, R., Powe, N.A., Turner, R.K., Bateman, I.J. and Langford, I.H. (1999). Public attitudes to contingent valuation and public consultation. *Environmental Values*, 8(3), 325-347.

Brouwer, R.; Langford, Ian H.; Bateman, I.J.; Crowards, T.C.; and Turner, R.K. (1997) A Meta-analysis of wetland contingent valuation studies. CSERGE Working Paper GEC 97-20

Burbridge, P.R.(1994). Integrated planning and management of freshwater habitats, including wetlands. *Hydrobiologia*, 285: 311-322.

Burgess, D., Hadley, D., Jackson, N., Turner, K., and Georgiou, S. (2004) The Tamar Catchment Study. Centre for Social and Economic Research on the Global Environment, School of Environmental Sciences, University of East Anglia.

Burgess, D.; Coombes, E.; Jackson, N.; Turner, R.K. and Cornell, s. (Forthcoming). Training Package and Policy Brief: Climate Change and Coastal Management in Practice – A cost-benefit assessment in the Humber, UK.

Caddy, J.F. (1993) Towards a comparative evaluation of human impacts on fishery ecosystems of enclosed and semi-enclosed seas. *Reviews in Fisheries Science*, 1, 57-95

Chichilnisky, G. and Heal, G. (1998) Economic returns from the biosphere. *Nature,* 391: 629-6300

Ciriacy-Wantrup, S.V. (1952) Resource Conservation: Economics and Policies. University of California Press, Berkeley.

Claridge, G.F. (1991) An overview of wetland values: a necessary preliminary to wise use. PHPA/AWB Sumatra Wetland Project Report No. 7, AWB, Bogor.

Clark, E.H. (1985) The off-site costs of soil erosion. *Journal of Soil and Water Conservation.* Jan-Feb: 19-21.

Colby, B. (1989) Estimating the value of water in alternate uses. *Natural Resources Journal*, 29 (2): 511-527.

Conlan, K. and Rudd, T., (2000). Sustainable Estuarine Development? Cumulative Impact Study of the Humber. *Journal of the Chartered Institution of Water and Environmental Management*, 14 (5): 313-317.

Coombes, E.G., (2003). *Habitat creation and loss within the Humber Estuary, and the associated environmental and economic costs and benefits*. M.Sc. by Research Dissertation, University of East Anglia.

Cooper, J.C.; Hanemann, M.; Signorello, G.; and Vigdor, J.L.. (2002) One-And-One-Half-Bound Dichotomous Choice Contingent Valuation *The review of Economics and Statistics* 84 (4) pgs 742-750

Costanza, R. (1994) Three general policies to achieve sustainability. In Jansson, A.-M., Hammer, M., Folke, C. and Costanza, R. (eds.) Investing in Natural Capital: The Ecological Economics Approach to Sustainability. Island Press, Washington, D.C.

Costanza, R., d'Arge, R., de Groot, R., Farber, S., Grasso, M., Hannon, B., Limburg, K., Naeem, S., O'Neill, R.V., Paruelo, J., Raskin, R.G., Sutton, P. and van den Belt, M., 1997. The value of the world's ecosystem services and natural capital. *Nature*, 387: 253-260.

Costanza, R., Farber, C.S. and Maxwell, J. (1989) Valuation and management of wetland ecosystems. *Ecological Economics*, 1: 335-361.

Cowardin, L.M., Carter, V., Golet, F.C., and LaRoe, E.T. (1979) Classification of wetlands and deepwater habitats of the United States. U.S. Fish and Wildlife Service, Washington, D.C.

Crooks, S., Turner, R.K., Pethick, J.S. and Parry, M.L., (2001). Managing catchment-coastal floodplains: the need for an UK water and wetlands policy. *Policy Analysis (PA) Working Paper* 01-01, Centre for Social and Economic Research on the Global Environment (CSERGE), University of East Anglia and University College London.

Crowards, T.M. (1996) Addressing uncertainty in project evaluation: The costs and benefits of safe minimum standards. Global Environmental Change Working Paper GEC 96-04, Centre for Social and Economic Research on the Global Environment (CSERGE), University of East Anglia and University College London.

Crowards, T.M. (1997) Combining economics, ecology and philosophy: Safe minimum standards of environmental protection. In O'Connor, M. and Spash, C. (eds.) Valuation and Environment: Principles and Practices. Edward Elgar, Aldershot, UK. forthcoming.

Cummings, R.G., Ganderton, P.T. and McGuckin, T. (1994) Substitution effects in CVM values. *American Journal of Agricultural Economics*, 76: 205-214.

De Groot, R.S (1992) Functions of Nature, Wolters-Noordhoff, Amsterdam.

Debinski, D.M. and Holt, R.D. (200) A survey and overview of habitat fragmentation experiments. *Conservation Biology* 14(2) 342-355.

DeBusk, (1999) *Nitrogen Cycling in Wetlands.* A fact sheet of the Soil and Water Science Department, Florida Co-operative Extension Service, Institute of Food and Agricultural Sciences, University of Florida.

Department for Environment, Food and Rural Affairs (DEFRA), (2000). *Flood and Coastal Defence Project Appraisal Guidance: Economic Appraisal. FCDPAG3*

Department for Environment, Food and Rural Affairs (DEFRA), (2003). *Flood and Coastal Defence Project Appraisal Guidance: Economic Appraisal: Supplementary Note to Operating Authorities. REVISIONS TO ECONOMIC APPRAISAL PROCEDURES ARISING FROM THE NEW HM TREASURY "GREEN BOOK"*.

Desvousges, W. H., Johnson, F. R., Dunford, R. W., Boyles, K. J., Hudson, S. P. and Wilson, K. N. (1992) Measuring non-use damages using contingent valuation: an experimental evaluation of accuracy. Research Triangle Institute Monograph 92-1, Research Triangle Park, NC: Research Triangle Institute.

Devon Biodiversity Partnership (1998) The Nature of Devon: A Biodiversity Action Plan – Rhôs Pasture. http://www.devon.gov.uk/biodiversity/pasture.shtml

Dickie, M., S. Gerking, M. Agee (1991): "Health benefits of persistent micropollutant control: the case of stratospheric ozone depletion and skin damage risks." In J.B. Opschoor and D.W. Pearce (eds), *Persistent Pollutants: Economics and Policy, Kluwer*, Dordrecht.

Dillman, D.A. (1978), Mail and Telephone Surveys – The Total Design Method, Chichester: John Wiley and Sons.

Dixon, A.M., Leggett, D.J. and Weight, R.C., (1998). Habitat Creation Opportunities for Landward Coastal Re-alignment: Essex Case Study. *Journal of the Chartered Institution of Water and Environmental Management*, 12: 107-112.

Dixon, J.A. (1989) Valuation of mangroves. Tropical Coastal Area Management Manila, 4(3): 2-6.

Dubourg, W.R. (1997). Reflections of the meaning of sustainable development in the water sector. *Natural Resources Forum*, 21 (3): 191-200.

English Nature (1991) Cornish Culm Grasslands: Botanical survey and conservation assessment 1989/90 Truro, UK: English Nature. http://www.wildlifetrust.org.uk/cornwall/wow/Reading/culm.htm

English Nature report (2001) Sustainable Flood Defence: The Case for Washlands (Report Number 406)

Environment Agency (2003) Project Apraisal Report: Knottingley Flood Alleviation Scheme.

Environment Agency (1997) Broadland flood alleviation strategy: Bank strengthening and erosion protection, Environment Agency, Suffolk.

Farber, S. and Costanza, R. (1987) The economic value of wetland systems, *Journal of Environmental Management,* 24(1): 41-51

Farnworth, E.G., Tidrick, T.H., Jordan, C.F. and Smathers, W.M. (1981) The value of ecosystems: An economic and ecological framework. *Environmental Conservation,* 8: 275-282.

Faux, J. and Perry, G.M. (1999) Estimating irrigation water value using hedonic price analysis: A case study in Malheur County, Oregon. *Land Economics* 75 (3) 40-452.

Fisher, A.C. and Krutilla, J.V. (1975) Resource conservation, environmental preservation, and the rate of discount. *Quarterly Journal of economics*. 89:358-370.

Folke, C. (1991) The Societal Value of Wetland Life-Support. In Folke, C. and Kaberger, T. (eds.) Linking the Natural Environment and the Economy: Essays from the Eco-Eco Group. Kluwer, Dordrecht.

Fordham, M. and Tunstall, S. (1990) The Trade-Off Between Flood Alleviation and Environmental Values. Publication given at the joint XII/Middlesex Polytechnic Workshop “Risk and Environmental Management”, Flood Hazard Research Centre, Middlesex Polytechnic.

Freeman, A.M. (1982) Air and Water Pollution Control. John Wiley, New York.

Garrod, G.D. and K.G. Willis (1994). Valuing biodiversity and nature conservation at a local level. *Biodiversity and Conservation* 3, 555-565.

Georgiou, S., Bateman, I., Cole, M. and Hadley, D. (2000) Contingent ranking and valuation of river water quality improvements: testing for scope sensitivity, ordering and distance decay effects. CSERGE Working Paper GEC 2000-18, Centre for Social and Economic Research on the Global Environment. University of East Anglia, Norwich, UK.

Gibbons, D.C. (1986) *The Economic Value of Water*. Resources for the Future, Washington D.C.

Gosselink, J.G., Odum, E.P., Pope, R.M., 1974. The Value of the Tidal Marsh.’ Center for Wetland Resources Publ.LSU-SG-74-03, Louisiana State University, Baton Rouge.

Green, C., Tunstall, S., Garner, J. and A.M. Ketteridge (1994) Benefits Transfer: rivers and Coasts, Paper presented for the CEBG meeting on benefits transfer, H.M. Treasury. Publication No. 231, Flood hazard Research Centre, Middlesex University.

Green, C.H. and Penning-Rowsell, E.C. (1986) Evaluating the intangible benefits and costs of a flood alleviation proposal. *Journal of Institution of Water Engineers and Scientist*, 40(3): 229-248.

Green, C.H. and Penning-Rowsell, E.C. (1989) Flooding and the quantification of 'Intangibles'. *Journal of the Institute of Water and Environmental Management*, 38:27-30.

Gren, I-M., Folke, C., Turner, R.K. and Bateman, I. (1994) Primary and secondary values of wetland ecosystems. *Environmental and Resource Economics*. Vol.4.: 55-74.

Grimble, R. and M.K. Chan. 1995. Stakeholder Analysis for Natural Resource Management in Developing Countries. *Natural Resources Forum* 19(2), 113-124.

Gutrich, J.J. and Hitzhusen, F.J. (2004) Assessing the substitutability of mitigation wetlands for natural sites: estimating restoration lag costs of wetland mitigation. *Ecological Economics* 28 (4) 409-424.

Hammack, J. and Brown, G.M., (1974) Water and wetlands: Towards bioeconomic analysis. Resources for the Future, Washington D.C

Hanley, N. D. (1990) Valuation of environmental effects: Final report - Stage One. Industry Department of Scotland and the Scottish Development Agency.

Hanley, N. D. and Craig, S. (1991) Wilderness development decisions and the Krutilla-Fisher model: the case of Scotland's Flow country. *Ecological Economics*, 4(2): 145-164.

Hanley, N. D. and Spash, C.L. (1993) Cost-benefit analysis and the environment. Edward Elgar, Aldershot.

Higgs, A.W. (1992) Purley on Thames Flood Alleviation Scheme. Paper to the Economic Appraisal Group, National Rivers Authority.

HM Treasury (1997) "The Green Book" Appraisal and Evaluation in Central Government. London: The Stationery Office.

Hocking, S., McCartney, P. (1999) An inventory of Culm Grassland in Cornwall. Vols. I and II. Truro, UK: Cornwall Wildlife Trust.

Holmes, T.P. and Adamowicz, W.L. (2003). Attribute-based methods. In: Champ, P.A., Boyle, K.J., Brown,T.C. (ed.s), A Primer on Non-market Valuation. Kluwer Academic Publishing Dordrecht.

Holmes, T.P., Bergstorm, J.C. Huszar, E., Kask, S.B., Orr III[e,] F. (2004) Contingent valuation, net marginal benefits, and the scale of riparian ecosystem restoration. *Ecological Economics* 49 (1), 19-20.

Holway, J.M. and Burby, R.J. (1990) The effects of floodplain development controls on residential land values. *Land Economics*, 66(3): 259-271

Hueting, R. (1980). New scarcity and economic growth, North Holland Publishing Company, Amsterdam.

Jickells, T., Andrews, J., Samways, G., Sanders, R., Malcolm, S., Sivyer, D., Parker, R., Nedwell, D., Trimmer, M. and Ridgway, J., (2000). Nutrient Fluxes through the Humber Estuary – Past, Present and Future. *Ambio*, 29 (3): 130-135.

Jordan, J. and Elnagheeb (1993) Willingness to pay for improvements in drinking water quality. *Water Resources Research*. 29(2):237-245.

Kahneman, D. and Knetsch, J. (1992) Valuing public goods: The purchase of moral satisfaction. *Journal of Environmental Economics and Management*, 22 (1): 57-70.

King, D.M. *et al.* (2000) Expanding Wetland Assessment Procedures: Linking Indices of Wetland function with Services and Values. ERDC/EL TR-00-17, US Army Engineer Research and Development Center, Vicksburg, MS.

King, A.W. (1993) Considerations of scale and hierarchy. In Woodley, S., Kay,J. and Francis, G., Ecological integrity and the management of ecosystems. St Lucie Press, Ottawa.

Kosz, M. (1996) Valuing riverside wetlands: The case of the “Donau-Auen” National Park. *Ecological Economics* 16: 109-127.

Krutilla, J.V. and Fisher, A.C. (1985) *The Economics of Natural Environments*, Johns Hopkins University Press, Baltimore.

Lahti, D.C. (2001) The “edge effect on nest predation” hypothesis after 20 years, *Biological Conservation* 99: 365-374.

Langford, I.H. and Bateman, I.J. (1999) Multilevel modelling and contingent valuation, in Bateman, I.J. and Willis, K.G. (eds.) *Valuing Environmental Preferences: Theory and Practice of the Contingent Valuation Method in the US, EU, and Developing Countries*, Oxford University Press, p442-459.

Langford, I.H., Bateman, I.J. and Langford, H.D. (1996) A multilevel modeling approach to triple-bounded dichotomous choice contingent valuation, *Environmental and Resource Economics*, 7(3):197-211.

Lareau, T. J. and Rae, D. A. (1989) Valuing WTP for diesel odor reductions: An application of the contingent ranking technique. *Southern Economic Journal*, 55 (3): 728-742.

Larson, L.A. (1986) Wetland and Flooding: Assessing Hydrologic Functions. In Kusler, J.A. and Riexinger, P. (eds.) Proceedings of the National Wetland Assessment Symposium, Association of State Wetland Managers, Portland Maine, June 17-20, 1985.

Ledoux, L., Cave, R. and Turner, R.K., (2002). The Use of Scenarios in Integrated Environmental Assessment of Coastal-Catchment Zones: the Humber Estuary, UK, *Land-Ocean Interactions in the Coastal Zone (LOICZ) Newsletter*, 23.

Ledoux, L., Brouwer, R. and Turner, R.K. (2000) Economic Appraisal of the Broads Fen Harvester Project. Final Report of the Life 97 ENV/UK/000511 New Wetland Harvests Final Technical Report.

Limburg, K.E., O'Neill, R.V. Costanza, R. (2002) Complex systems and valuation. *Ecological Economics* 41 (3), 409-420.

Loomis J.B. (1987a) The economic value of instream flow: Methodology and benefit estimates for optimum flows. *Journal of Environmental Management*, 24: 169-179.

Loomis J.B. (1987b) Balancing public trust resources of Mono Lake and Los Angeles' water right: an economic approach. *Water Resources Research* 23(8): 1449-1456.

Maltby, E. and Immirzi C.P. (1993). Carbon dynamics in peatlands and other wetland soils - regional and global perspectives. *Chemosphere*, 27 (6), 999 - 1023.

Maltby, E., Holdgate, M., Acreman, M., Weir, A., (1999), Ecosystem Management: Questions for Science and Society, Royal Holloway Institute for Environmental Research, Virginia Water, UK.

Margat, W., W. Viscusi, J. Huber (1987): "Paired comparisons and contingent valuation approaches to morbidity risk valuation." *Journal of Environmental Economics and Management*, **15**, 1987.

Marin, A. and Psacharopoulos, G. (1982) The reward for risk in the labor-market – evidence from the United-Kingdom and a reconciliation with other studies. *Journal of Political Economy*. 90(4) 827-853.

Markandya, A. and Pearce, D.W. (1988) Environmental Considerations and the Choice of Discount Rates in Developing Countries. Environment Department Working Paper No. 3, World Bank, Washington DC.

Mitchell, B. (1990) Integrated Water Management: International Experiences and Perspectives, Belhaven Press. London.

Mitchell, R.C. and R.T. Carson (1989) Using Surveys to Value Public Goods: The Contingent Valuation Method. Resources for the future. Washington D.C.

Mitsch, W.J., and Gosselink, J.G. (2000) Wetlands. 3rd edition. Wiley. New York.

Möller, I., Spencer, T., French, J.R., Leggett D.J. and Dixon, M., (2001). The sea-defence values of salt marshes: Field evidence from north Norfolk. *Journal of the Chartered Institution of Water and Environmental Management* 15 (2): 109-116.

Morey, E. R. (1984) The choice of ski areas: estimation of a generalised CES preference ordering with characteristics. *Review of Economics and Statistics*, 66: 584-590.

Morey, E. R. (1985) Characteristics, consumer surplus, and new activities: a proposed ski area. *Journal of Public Economics*, 26 (2): 221-236.

Morgan, M.G. and Henrion, M. (1990) Uncertainty: A Guide to Dealing with Uncertainty in Quantitative Risks and Policy Analysis, Cambridge University Press, Cambridge.

Moss, B. (1998) The E numbers of eutrophication – Errors, ecosystem effects, economics, eventualities, environment and education. *Water Science Technology* 37 (3) 75-84.

N'Jai, A., Tapsell, S.M., Taylor, D., Thompson, P.M., Witts, R.C., Parker, D.J., Penning-Rowsell, E.C. (1990) Flood Loss Assessment Information Report. Flood Hazard Research Centre, Middlesex University, UK.

Naeser, R.B. and Bennett, L.L. (1998) The Cost of Noncompliance: The Economic Value of Water in the Middle Arkansas River Valley. *Natural Resources Journal,* vol. 38(3): 445-463.

National Research Council (1997) Valuing Ground Water. Economic Concepts and Approaches. Committee on Valuing Groundwater. Water Science and Technology Board. Commission on Geosciences, Environment and Resources. National Research Council. National Academy Press. Washington DC.

National Rivers Authority, (1992) A flood alleviation strategy for Broadland: final report annex four – Cost Benefit Studies, NRA, Anglian Region, Peterborough.

Norton, B.G. (1986) Towards Unity Among Environmentalists. Oxford University Press, Oxford.

O'Neil, J. (1997) Managing Without Prices: The Monetary Valuation of Biodiversity. *Ambio*, 26; 546-550.

O'Riordan, T. and Ward, R. (1997) Building Trust in Shoreline Management: Creating Participatory Consultation in Shoreline Management Plans. *Land Use Policy*, 14(4): 257-276.

O'Riordan, T., Andrews, J.E., Samways, G., and Clayton, K., (2000). Coastal Processes and Management. In: *Environmental Science for Environmental Management*, O'Riordan, T., (ed.), (second edition), p.243-266, Prentice Hall, Harlow, UK.

Parkes, D. (2003) Storage and cycling of organic carbon and nutrients in Holocene coastal sediments. Ph.D. Thesis, UEA.

Pearce, D.W., Markandya, A. and Barbier, E.B. (1989) Blueprint for a Green Economy. Earthscan, London.

Pearce, D.W.and Turner, R.K.(1990) Economics of Natural Resources and the Environment, Harvester Wheatsheaf, London.

Pethick, J., (2001). Coastal management and sea-level rise. *Catena* 42: 307-322.

Pilcher, R., Burston, P. and Davis, R., (2002). *Seas of Change*, A RSPB report.

Poe, G. (1998) Valuation of groundwater quality using a contingent valuation-damage functions approach. *Water Resources Research*, 34 (12): 3627-3633.

Reed, D.J., Spencer, T., Murray, A.L., French, J.R. and Leonard, L., (1999). Marsh surface sediment deposition and the role of tidal creeks: implications for created and managed coastal marshes. *Journal of Coastal Conservation*, 5: 81-90.

Renzetti, S. (2002) *The Economics of Water Demands,* Norwell, MA: Kluwer Academic Press.

RPA. 1998. *The environmental costs and benefits of water resources*, report prepared for the Environment Agency.

Ruttan, V. (1965) The economic demand for irrigated acreage: new methodology and some preliminary projections, 1954- 1980, Resources for the Future, Baltimore.

Sagoff, M. (1988). *The economy of the earth*. Cambridge University Press. Cambridge

Sagoff, M. (2004). *Price, Principle and the environment*, Cambridge University Press, Cambridge.

Sather, J.H. and Smith, R.D. (1984) An Overview of Major Wetland Functions and Values. FWS/OBS-84/18, Fish and Wildlife Service: US Dept of the Interior.

Saunders, D.A., Hobbs, R.J., Margules, C. R. (1991) Biological consequences of ecosystem fragmentation: a review, *Conservation Biology* 5: 18-32.

Saunders, G. (2003) Culm Grassland: A Review and Vision to the Year 2013. A report on behalf of Devon Wildlife Trust For English Nature

Smith, V. K. (1992) Arbitrary Values, Good Causes, and Premature Verdicts. *Journal of Environmental and Resource Economics*, 22 (1): 71-89.

Smith, V. K. and Desvousges, W. H. (1986) Measuring Water Quality Benefits. Boston: Kluwer-Nijhoff Publishing.

Smith, V. K. and Kaoru, Y. (1990). Signals or Noise? Explaining the Variation in Recreation Benefit Estimates," *American Journal of Agricultural Economics* 72 (2): 419-433.

Sorg, C.F. and Loomis, J.B. (1984) Empirical Estimates of Amenity Forest Values: A Comparative Review. General Technical Report, RM-107, Rocky Mountain Forest and Range Experiment Stateion, Forest Service, USDA, Fort Collins CO.

Steele, J.H (1991) Marine Functional Diversity. *Bioscience*, 41:470-474.

Thaler, R. (1984) Toward a positive theory of consumer choice. *Journal of Economic Behaviour and Organisation,* 1: 29-60.

Tobin, G.A. and Newton, T.G. (1986) A theoretical framework of flood induced changes in urban land values. *Water Resources Bulletin*. 22(1): 67-71.

Tunstall, S.M., Tapsell, S.M. and Fordham, M. (1994) Public Perception of Rivers and Flood Defence: Final Report. R&D Note 444, National Rivers Authority, Bristol.

Turner R.K., van der Bergh, J.C.J.M., Berendregt, A. and Maltby, E. (1997) Ecological-economic analysis of wetlands: Science and social scfience integration. In: Soderquist, T. (ed.), Wetlands: Landscape and institutional perspectives. Proceedings of the 4th workshop of the Global Wetlands Economics Natwork (GWEN). Beijer International Institute of Ecological Economics, The Royal Swedish Academy of Sciences, Stockholm, Sweden, 16-17 November 1997.

Turner, R.K. & Brooke, J. Management and Valuation of an Environmentally Sensitive Area: Norfolk Broadland Case Study, *Environmental Management*, Vol. 12, No. 2, pp.193-207, May 1988.

Turner, R.K. (1993). Sustainable Environmental Economics and Management: Principles and Practice. Belhaven Press, London and New York.

Turner, R.K. (2000) Integrating natural and socioeconomic science in coastal management. *Journal of Marine Sciences*, 25: 447-460.

Turner, R.K. and Pearce, D.W. (1993) Sustainable Economic Development: Economic and Ethical Principles. In Barbier, E.B. (eds.) Economics and Ecology: New Frontiers and Sustainable Development. Chapman and Hall, London.

Turner, R.K. and Pearce, D.W. (1993) Sustainable Economic Development: Economic and Ethical Principles. In Barbier, E.B. (eds.) Economics and Ecology: New Frontiers and Sustainable Development. Chapman and Hall, London.

Turner, R.K., Dent, D. and Hey, R.D. (1983) Valuation of the environmental impact of wetland flood protection and drainage schemes. *Environment and Planning A*, 15:871-888.

Turner, R.K., Doktor, P. and Adger, N. (1995). Assessing the costs and benefits of sea level rise. *Environment and Planning A* 27: 1777-1796.

US Environmental Protection Agency (1985) *Costs and Benefits of Reducing Lead in Gasoline: Final Regulatory Impact analysis.* Washington DC: EPA-230-05-85-006.

Weatherhead, E.K. and Knox, J.W. (2000) Predicting and mapping the future demand for irrigation water in England and Wales. *Agricultural Water Management*. 43(2) 203-218.

West, J.J. and Dowlatabadi, H, 1999: Assessing economic impacts of sea level rise. In: Climate Change and Risk [Downing, T.E., A.A. Olsthoorn, and R. Tol (eds.)]. Routledge, London, United Kingdom, pp. 205–220.

Wigley, T.M.L. and Raper, S.C.B, (1993) Future changes in global mean temperature and sea level. In: Climate and sea level change: observations, projections and implications, Warrick, R.A., E.M. Barrow and T.M.L.Wigley (eds) Cambridge University Press, Cambridge, UK.

Willis, K. G. and J. F. Benson (1988): "A Comparison of User Benefits and Costs of Nature Conservation at Three Nature Reserves". *Regional Studies* 22: 417-28.

Winn, P.J.S., Young, R.M., and Edwards A.M.C., (2003). Planning for the rising tides: the Humber Estuary Shoreline Management Plan. *The Science of the Total Environment*, In Press.

Yohe G, Neumann J, (1997) Planning for sea level rise and shore protection under climate uncertainty. *Climatic Change* 37 (1): 243-270

Yohe, G, Neumann, J, Ameden, H. (1995). Assessing the economic cost of greenhouse induced sea level-rise – methods and application in support of a national survey. *Journal of Environmental Economics and Management*. 29 (3): S78-S97 Part 2 Suppl. S

Young, R.A. (1996) Measuring economic benefits for water investments and policies. World Bank Technical Paper No. 338.

Young, R.A. and Gray, S.L. (1972) Economic Value of Water: Concepts and Empirical Estimates. Technical Report to the National Water Commission.

Annex A: Examples of wetland characteristics

- Size
- Shape
- Species presence/abundance/rarity
- Vegetation structure
- Patterns of vegetation distribution
- Soils
- Geology
- Geomorphology
- Processes (biological, chemical and physical)
- Nature and location of water entry and water exit
- Climate
- Location in respect of human settlement and activities
- Location in respect of other elements in the environment
- Water flow/turnover rates
- Water depth
- Water quality
- Altitude
- Slope
- Fertility
- Nutrient cycles
- Biomass production/export
- Habitat type
- Area of open water
- Recent evidence of human usage
- Historic or prehistoric evidence of human usage
- Tidal range/regime
- Characteristics of the catchment

Source: Claridge (1991)

Annex B. Extended Catchment Spatial Scale

In the context of water resources management, Mitchell (1990) has argued that efforts towards a more integrated water resources management has three related dimensions. In the case of the management of wetlands, integration can be interpreted as follows:

- **in systems ecology terms**, i.e., how each component of the wetland system (at the catchment scale) influences other components;
- in **wider biogeochemical and physical systems terms**, i.e., where water interacts with other biophysical elements (one of the most characteristic features of a wetland);
- in **socio-economic, socio-cultural and political terms**, i.e., the linkage of wetland management to relevant policy networks and economic and social systems (with attendant culture and history) so that chances of achieving a co-operative solution or mitigation strategy are maximised.

Management of wetlands is therefore intimately connected with an appreciation of the full functioning of the hydrological, ecological and other systems and the total range of structures and processes and functional outputs of goods and services that are provided. Wetland management and pricing must therefore be based on a relatively wide (at least catchment scale) appreciation of the landscape ecological processes present, together with the relevant environmental and socio-economic driving forces. In order to manage wetlands holistically, one of the primary issues is whether the scale of administrative structures and appropriately refined scientific support equate with the scale of catchment processes. Management of wetlands and water resources more generally is all too often focussed on a sectoral basis, and constrained by political and institutional considerations. The proprietorial interests shown by communities towards their localities in catchments are extremely powerful forces, which democratic systems often find difficult to accommodate. Yet wetland ecosystems are driven by hydrological and ecological processes that transcend the local scale and the short term. These linked hydrological-ecological systems provide a wide range of benefits and services that are often ignored or under-valued in water-use planning, leading to their long term loss.

Under a catchment wide perspective, interrelationships are made explicit and provide an important basis for decision making involving multiple wetland users. For instance, water abstraction or water pollution upstream may have severe consequences downstream. The important issue of the distribution of costs and benefits of (changes in) wetland use only becomes visible if considered at their appropriate scale in time and space. For wetland ecosystems this is the catchment level, without which, it will be difficult to trace the impact of any upstream user's decision on the downstream beneficiaries of the service. Thus it is difficult to allocate the value of the service and include it in the decision making process. Furthermore, many wetland values can only be realised if a minimum of upstream users take the catchment perspective into account in their decision. For example, in a given area, a minimum amount of land users may have to agree to maintain riparian buffer strips to guarantee a certain water quality for domestic use for a downstream city, which makes negotiations complex. In such cases where land uses have a noticeable impact on downstream water values, the land property rights will be just as important to take into account in valuation exercises as the water property rights. Furthermore, economic analysis is not limited to areas with functional linkages to the wetland, but is generally more concerned with the economic region of influence and the range of relevant stakeholder interests and positions. This may roughly conform to patterns of the local physical environment but is by no means determined by it.

Annex C: Total Economic Value and Other Natural Values

The approach that is advocated in this report describes the components of the value of wetlands using the conventional categories of Total Economic Value (Figure C1). There are two main categories, use values and non-use values:

- Direct use values arise from direct interaction with wetlands. They may be consumptive, such as use of water for irrigation, the harvesting of fish or they may be non-consumptive such as recreation, or the aesthetic value of enjoying a view. It is also possible that 'distant use' value can be derived through the media (e.g. television and magazines), although the extent to which this can be attributed to a specific site, and the extent to which it is actually a use value, are unclear.
- Indirect use values are associated with services that are provided by wetlands but that do not entail direct interaction. They are derived, for example, from flood protection provided by wetlands or the removal of pollutants by aquifer recharge.

<table>
<tr><td rowspan="9">TOTAL ECONOMIC VALUE</td><td rowspan="4">Use value</td><td>Consumptive use value</td></tr>
<tr><td>Recreational, aesthetic and educational use value</td></tr>
<tr><td>Distant use value</td></tr>
<tr><td>Indirect use value</td></tr>
<tr><td rowspan="3">Non-use value</td><td>Existence value</td></tr>
<tr><td>Bequest value</td></tr>
<tr><td>Philanthropic value</td></tr>
<tr><td rowspan="2"></td><td>Option value</td></tr>
<tr><td>Quasi-option value</td></tr>
</table>

Figure C1: Components of the Total Economic Value of Wetlands

Non-use values are derived from the knowledge that a resource is maintained. By definition, they are not associated with use of the resource or tangible benefits that can be derived from it (though resource users may derive non-use values). Non-use values are linked to ethical concerns and altruistic preferences, though it can be argued that these may ultimately stem from self-interest. They can be divided into three types of value (which can be overlapping): existence value, bequest value and philanthropic value. Existence value is the satisfaction derived from knowledge that a feature of a wetland continues to exist, regardless of whether or not it might be of benefit to others. Bequest value is derived from the knowledge that a feature of a wetland will be passed on to future generations so that they will have the opportunity to enjoy it in the future. Finally, philanthropic value is the satisfaction gained from ensuring that resources are available to contemporaries in the current generation.

There are two further types of value that are not categorised as either use or non-use values. These are option value and quasi-option value:

- Option value is the satisfaction that an individual derives from the ensuring that a resource is available for the future given that the future availability of the resource is uncertain. It can be regarded as insurance for possible future demand for the resource.
- Quasi-option value is derived from the potential benefits of waiting for improved information prior to giving up the option to preserve a resource for the future. This is based on a desire to take advantage of the prospect of improved information in the future and act on subsequent revision of preferences. It is the value placed on retaining flexibility, and avoiding irreversible damage that might prove to be undesirable in the light of future information. An example is the value placed on conservation of a wetland until further information is available on the value of the species that are found within it.

Total economic value is determined as the sum of the components in Figure C1. In practical terms this is limited to those components that it is feasible to quantify. Use of total economic value in the analysis of alternative allocations ensures that the full social benefit of goods and services provided by wetlands are taken into account. This is necessary to indicate to decision-makers the welfare improvement that is offered by alternative allocations. Total economic value does not, however, provide an exhaustive assessment of the value of wetlands to society. It measures the extent to which goods and services provided by wetlands touch on the welfare of society, as direct determinants of individuals' wellbeing or via production processes. It represents two fundamental sets of values: individual values and production values. Individual values include recreational and amenity values, as well as non-use values (existence, bequest and philanthropic values) of goods and services provided by wetlands. Production values occur through the influence of wetlands on the production and cost functions of other marketed goods and services (such as use of water as an intermediate good in irrigated crop production). The effects of this influence on the prices of other inputs and marketed goods and services translate into changes in individuals' welfare.

However, as was pointed out earlier in this report, there is another set of values that is supplementary to total economic value. This represents the role of wetlands in natural systems. It includes the value of services that stabilise natural systems and perform protective and supportive roles for economic systems. These values are more usually presented in relation to biodiversity. They include the following (somewhat overlapping) categories of value which have appeared in the scientific literature:

- inherent value: the value of those services without which there would not be the goods and services provided by the system (Farnworth et al., 1981);
- contributory value: this represents the economic-ecological importance of species diversity. Species that are not of use to humans are important because they contribute to increased diversity, which itself contributes to the generation of more species (Norton, 1986);
- indirect use value: this is related to the support and protection that is provided to economic activity by regulatory environmental services (Barbier, 1994);

- primary value: incorporates the fact that existence of the ecosystem structure is prior to the range of function/good and service values (Turner and Pearce, 1993);
- infrastructure value: this relates to a minimum level of ecosystem 'infrastructure' as a contributor to its total value (Costanza et al, 1997).

These values build on three important aspects of the ecology of natural systems:

1. **Complementary relationships.** Species co-exist within natural systems, defined by complex relationships of interaction and interdependence. Survival of one species depends on the existence of other species, which in turn depend on others. This 'contributory value' focuses on the survival of species within the web of interactive relationships with each species contributing to the survival of others. Contributory value is based on the limited substitutability of species. This occurs because every species performs very specific duties within the ecological system. The role of contributory values is not usually taken into account explicitly, because the required knowledge (on ecological interrelationships) is unavailable, but it can be incorporated through adoption of a precautionary approach to resource management.

2. **Keystone species.** The persistence of natural systems in their current existing states may be dependent on a limited number of biotic and physical processes. These processes are directed by groups of species with complementary functions, known as "keystone species". Other species are redundant, though can become keystone species under a change in environmental conditions. As long as species can substitute for each other under changing conditions, the balance of processes within the system can remain in tact. Reductions in the diversity of species in the system do, however, diminish the possibilities for substitutions under a change in conditions. This limits the capacity of the system to persist in its current state in the face of stresses and shocks. More recently ecologists have highlighted the importance of keystone processes rather than individual species.

3. Goods and services provided by a natural system are dependent on the **structure and functioning of the systems**. The goods and services provided are inherently connected to the integrity of the natural system and the totality of the structure and functioning of the system (Farnworth et al, 1981). This can be understood in terms of the concepts of primary and secondary value (Gren et al, 1994). The primary value describes the system characteristics: the self-organising capacity of the system including its dynamic evolutionary processes and capacity to absorb external disturbances. It relates to the aspects of the system that 'hold everything together' and is consequently also referred to as 'glue value'. Secondary value refers to the renewable flow of goods and services generated by the natural system. It is dependent on the continued operation, maintenance and 'health' of the system as a whole.

Total economic value does not give credit to this set of values and, therefore, is not exhaustive. Such values are particularly relevant to single function natural systems, the contributory value of which can only be properly addressed when the site is viewed within the context of the larger catchment system. The recognition of complementary relationships implies that the total value of ecosystems is infinite. This is similar to the consideration of wetlands and water resources as a form of "critical" natural capital (Dubourg, 1997). Here again, the value of these resources is infinite and the usual measures of value (market price; willingness to pay) do not reflect the true economic value of the resource. As a basis of human life, complementary relationships with wetlands are indispensable under realistic technological and economic conditions. However, apparently marginal decisions (as perceived by different stakeholders) are the stuff of the real world and therefore need to be considered. The problem is that knowledge about the consequences of resultant infringements on natural systems is incomplete. There is an unbridgeable gap in knowledge about natural system interrelationships and regularities. The benefits of protection will often only be discovered once the natural system has been disturbed or lost.

Socio-cultural and Historical Dimensions of Natural Values

The task of sustainable management can be defined as sustainable utilisation of the multiple goods and services generated by natural systems, together with 'socially equitable' distribution of welfare gains and losses inherent in such usage. However, social welfare is affected not only by changes in economic welfare but also changes in properties of natural resources that are associated with people's sense of identity, their culture and which are of historical significance. Such properties are particularly important in the case of wetlands, given the essential role of water for human life. The compilation of data for such properties is a qualitative exercise, involving more deliberative and inclusionary interest group approaches such as consensus conferences, citizen juries and focus group interviewing. Different cultural views on social relations are assumed to give rise to different degrees of support for alternative decision–making procedures and for the underlying valuations elicited via the social discourse process (O'Riordan and Ward, 1997; Brouwer et al, 1999). This has similarities to the so-called 'approved process' approach (Morgan and Henrion, 1990) in which all relevant parties observe a specified set of procedures or concept of due process to make a decision that balances conflicting values at the political level.

Some environmental analysts claim that natural systems also have non-anthropocentric intrinsic value and that non-human species possess moral interests or rights, or that though all values are anthropocentric and usually (but not always) instrumental, the economic approach to valuation is only partial. These environmentalist positions lead to the advocacy of environmental sustainability standards or constraints, which to some extent obviate the need for valuation of specific components of the environment. It is still necessary, however, to quantify the opportunity costs of such standards, or to quantify the costs of current, and prospective environmental protection and maintenance measures – see figure 4. Nevertheless, some commentators view it as feasible and desirable to manage the environment without prices. For example, O'Neil (1997) found that in other arenas such as forestry and biodiversity management, issues concerning conflicts in value are resolved through pragmatic methods of argument between botanists, ornithologists, zoologists, landscape managers, members of the local community, and farmers.

There is a growing body of evidence that suggests that some of the conventional economic axioms are systematically violated by humans in controlled experiments and in everyday life. To take just one issue, it seems likely that individuals recognise 'social interest' and hold social preferences separate from self-interested private preferences. The origin of this social interest may be explained by theories of reciprocal altruism, mutual coercism, or by sociobiological factors. The distinction between the individual as a citizen and as a consumer is not, therefore, an either/or issue, but is more properly interpreted as the adoption of multidimensional roles by individuals.

As citizens, individuals are influenced by held values, attitudes, and beliefs about public goods and their provision. In this context, property rights (actual and/or perceived), social choices and moral concerns can all be involved in the conflict between conservation and development of natural resources. The polar view to the conventional economic approach holds that the very treatment of ecological assets such as biodiversity in terms of commercial norms is itself part of the environmental crisis. The argument becomes one of the 'proper' extent of market influences and commodification (O'Neil, 1997). Advocates of this perspective argue that market boundaries should not be extended to cover as many environmental assets as possible. Instead society should give greater consideration to the nature of deliberative institutions for resolving environmental problems and the social and economic framework that sustains them (O'Neil, 1997). A counterbalancing argument is that some environmental goods and services that have mixed public and private good characteristics (e.g. forests, catchments, areas with ecotourism potential and some aspects of biodiversity services) could be privatised or securitised (shares issued). In this way self-interest and the profit motive can be made to work in favour of environmental conservation (Chichilnisky and Heal, 1998).

Annex D: Economic Valuation Techniques

Direct valuation techniques

Direct valuation techniques seek to directly elicit preferences through questioning individuals on their willingness-to-pay for a good or a service. These techniques include the Contingent Valuation Method (CVM), contingent ranking and conjoint analysis. The most common direct valuation approach is the Contingent Valuation Method.

Contingent Valuation Method

The Contingent Valuation Method can be useful for several types of problems including, water quality, recreation sites, option and existence values of biodiversity, and water. CVM can be used to calculate both use and non-use values including option and existence values. The contingent valuation method uses a survey instrument to measure individuals' maximum willingness to pay (WTP) for natural resources presented to them in an hypothetical market with a proposed improvement (Hanley and Spash, 1993). It can also be used to measure what people are willing to accept (WTA) by way of compensation for a change in the quality of a good.

People are asked questions regarding the amount of money they would be willing to pay for an improvement in the environmental good or service in question through face to face interviews, telephone or mail surveys. However, in developing countries, face to face interviews are considered as the most appropriate way to carry out the survey since the rate of illiteracy is high and telephonic network often defective. The design of the questionnaire is important and must include in a first part, an explanation of what the environmental problem is together with information on the environmental change, questions regarding the WTP or WTA in a second part. The third part of the questionnaire will consist of questions about the socio-economic characteristics of the interviewee, which will enable to analyse and verify the validity of the answers on the WTP or WTA given by the respondents.

A respondent's choice or preference can be elicited in a number of ways. The simplest is to ask the respondent a direct question about how much he or she would be willing to pay for the good or service – known as continuous or open-ended questions. High rates of non-responses can be a problem with this approach. Alternatively, a respondent can be asked whether or not he or she would want to purchase the service if it cost a specified amount. These are known as discrete or dichotomous choice questions, and may be favoured because they do not give the respondent any incentive to answer untruthfully, that is the approach is 'incentive compatible'. A hybrid approach is the 'bidding game', where respondents are asked a series of questions to iterate towards a best estimate of their valuation. Alternatively, respondents may be shown a list of possible answers – a 'payment card' – and asked to indicate their choice, though this requires a careful determination of the range of possible answers. Each approach implies particular requirements in terms of statistical methods, and the appropriate choice for a specific problem is a matter of judgement on the part of the analyst.

One of the problems with the application of the CVM, is that it is subject to biases. The problem of strategic bias has long worried economists. The likelihood of the occurrence of strategic behaviour depends on the respondent's perceived payment obligation and his or her expectation about the provision of the good. Where individuals believe they will actually have to pay their reported WTP, but that their personal valuation will not affect whether the good is provided or not, there is a temptation to understate the true value in the hope of a 'free-ride', that is that others will pay. If, however, the price to be charged for the good is not tied to an individual's WTP response, whereas provision of the good is, then over-reporting of WTP might occur in order to ensure provision. Incentive compatible payment methods might minimise the risk of strategic behaviour. Overall, fears of strategic bias problems have not been substantiated by the large amount of empirical investigation into the question.

Hypothetical bias – caused by the hypothetical nature of the CV market – could mean that respondents' answers are meaningless if their declared intentions cannot be taken as an accurate guide to their actual behaviour. This is most likely to occur if respondents are very unfamiliar with the scenario presented to them. A careful and believable description of the good and its context can help in this instance. A survey of experimental tests, which compare hypothetical bids with those obtained in simulated markets where real money transactions take place, suggests that hypothetical bias can be reduced significantly if WTP formats are used instead of WTA, the reason being that respondents have more practical experience with payment than with compensation scenarios (Hanley, 1990).

Analysts will often wish to summarise respondents' valuation estimates in terms of the mean WTP for the good or service, or to develop an aggregate benefit estimate for a community or region. Two types of problems, which might produce 'aggregation bias', involve sampling errors and insufficient sample sizes. Sampling errors might arise because survey non-responses are more likely to occur for certain types of individuals who are not randomly distributed in the population, resulting in a non-random survey sample. Similarly, if sample size is small, there is a risk that the characteristics of the sample will not be representative of the general population.

A number of studies have found evidence of 'payment vehicle bias', where WTP depends upon the choice of the method of payment, for instance, between tax increases or entrance fees. Controversial payment vehicles should be avoided in favour of those most likely to be employed in real life to elicit payment for the good in question. But the fact that the respondents' answers may depend on precisely how they are asked to pay for the hypothetical good or service should be expected; it should not be a source of concern because a preference for one payment vehicle over another may be perfectly reasonable. In this sense the term 'bias' is misplaced.

Starting point bias arises when the initial value suggested at the beginning of a bidding game has a significant impact upon the final bid reported by the respondent. The use of starting points can reduce valuation variance and the number of non-responses in open-ended type questionnaires, but at the possible expense of respondents not giving serious thought to their answers and taking cognitive 'short cuts' in arriving at their decision. One solution might be to use a 'payment card', with a range of numbers from which respondents can select their bid. However this can result in an 'anchoring' of bids within the range presented. It has been argued that an optimal range of prices should include a low price that results in almost all respondents accepting it, and a high price that results in almost all respondents rejecting it. Within this range, prices offered should reflect the distribution of bids so that, optimally, each bid interval reflects the same proportion of the population (Bateman *et al.*, 1992).

Perhaps most controversy has centred on the so-called 'embedding effect'. A few studies have found that individuals' CV responses often do not vary significantly with changes in the scope and coverage of the environmental good being valued (Kahneman and Knetsch, 1992; Desvousges *et al.*, 1992). In these studies respondents appear not to discriminate between the particular environmental good under consideration, and the general class of environmental goods it belongs to. A number of explanations has been advanced for this phenomenon. Some have argued that it is because individuals' do not possess strongly articulated preferences for environmental goods, so that they tend to focus on other facets of the environment, such as the 'moral satisfaction' associated with the preservation of particular species or habitats (Kahneman and Knetsch, 1992) when deciding on a monetary valuation. Others have argued that embedding is more an artefact of poor survey design (for example Smith, 1992). Another suggestion is that, to make valuation and financial decisions easier, people think in terms of a system of expenditure budgets, or 'mental accounts', to which they allocate their income (Thaler, 1984). If the amount allocated to the 'environment account' is quite small, then this might result in an inability to adjust valuations substantially in response to changes in the size and scope of an environmental good. Essentially, embedding might be a result of valuations' being determined by an income constraint, which is inflexible and relatively strict, compared with assessments of an individual's total (or full) income.

The debate over embedding has not yet been resolved. Whether the effect is a robust one or not, it does appear that its severity can be reduced by careful survey design, and in particular by giving precise, contextual descriptions of the good itself, and of the expenditure implications of a particular WTP bid.

There are numerous issues that arise in CV work in developing countries that demand careful attention in order to increase the probability that high-quality results are obtained. The first difficulty is to explain to interviewers what the study is about and especially the concepts of economic value and maximum willingness to pay or minimum willingness to accept. Another problem is the presumed difficulty of understanding and interpreting respondents answers to sensitive or hypothetical questions.

The US National Oceanic and Atmospheric Administration (NOAA) panel has offered a set of guidelines it believes should be followed if CV is to provide information about non-use values of sufficient quality as to be usable as the basis for claims for legal compensation for environmental damage (Arrow *et al.*, 1994). The use of these guidelines within the profession is now being extended to cover all CV studies. CV is likely to be most reliable for valuing environmental gains, particularly when familiar goods are considered, such as local recreational amenities. Finally, it should be remembered that CVM is the only technique with the potential for measuring existence value.

Contingent Ranking

Contingent ranking is implemented in the same vein as contingent valuation except that the respondent in the experiment is asked to rank order a large number of alternatives with various combinations of environmental goods and prices. A random utility framework is then used to analyse the data on the complete ranking of all the alternatives. The statistical estimation is often done with what is essentially a multinomial logit model of the rank order of the random utility level associated with each alternative. Implicit attribute prices or welfare change measures are then calculated from the parameter estimates of this logit model.

Applications of contingent ranking have usually involved the ranking of large numbers of alternatives, which often appear similar to the respondent. As such, the cognitive task of arriving at a complete ranking is found to be very difficult. Furthermore, the estimated statistical models are often poor and result in imprecise environmental values. The contingent ranking method has therefore met with a mixed response (Smith and Desvousges; 1986, Lareau and Rae, 1989).

Conjoint Analysis

Conjoint analysis has strong foundations in psychology and statistics, but is rather shaky on theoretical foundation from the point of view of individual choice theory. However, there is a trend in valuation studies to move away from reliance on purely statistical methods towards more behaviourally based models such as multinomial logit model. One of the features of conjoint analysis is that each individual participating in the conjoint analysis experiment is faced with a large number of ranking tasks. Each ranking task involves a small number of alternatives, for example, two. Then based on the collected data, some type of utility index model is estimated for one individual. This differs from contingent valuation and ranking where a large number of individuals are asked about their stated preferences for one set of alternatives, and a representative random utility model is estimated for the relevant population.

Indirect valuation techniques

Indirect approaches rely on observed behaviour to deduce values. Indirect valuation techniques include, the travel cost method, hedonic pricing approach, averting behaviour method, dose-response technique, replacement cost method, market valuation, simulation models, optimization models and other methods.

Travel Cost Method

Many natural resources, such as lakes and forests, are used extensively for the purpose of recreation. But it is often difficult to value these resources because no prices generally exist for them with which to estimate demand functions. However, travel cost models take advantage of the fact that, in most cases, a trip to a recreation site requires an individual to incur costs in terms of travel and time. Different individuals must incur different costs to visit different sites, and these ‘implicit’ prices can be used in place of conventional market prices as the basis for estimating the value of recreation sites and changes in their quality. Clearly, because travel cost models are concerned with active participation, they measure only the use value associated with any recreation site – option and existence values must be estimated via some other technique.

There are two variants of the simple travel cost visitation model. The first can be used to estimate (representative) individuals’ recreation demand functions. This is done by observing the visitation rate of individuals who make trips to a recreational facility, as a function of the travel cost. The value of a recreation site to an individual is measured by the area under his or her demand curve, so that the total recreation (use) value of a site is simply the area under each demand curve summed over all individuals. This ‘individual’ travel cost model requires that there is variation in the number of trips individuals make to the recreational site, in order to estimate the demand function. One particular problem therefore arises from the fact that such variation is not always observed, especially since all individuals do not always make a positive number of trips to a recreational site. Some individuals do not make any. This means that if we are to use standard statistical techniques such as Ordinary Least Squares (OLS) for the analysis of such models, we must exclude non-participants from our data sample. This will not only imply probably very low participation rates, but also a loss of potentially useful information about the participation decision. However including all individuals in a sampling area does mean that more complex statistical methods are required – in particular, discrete choice models.

Alternatively, a simpler solution is to model the demand for a site as an aggregate or market demand, using the standard statistical techniques. This second variant of simple travel cost models is known as the ‘zonal’ travel cost method. The unit of observation is now the ‘zone’, as opposed to the individual. The visitation rate used is the number of trips per capita from each zone. Zones are constructed by dividing the region around a site into areas of increasing travel cost. The observations of trips are then allocated to their ‘zone of origin’, and the population of each zone is found. The visitation rate is calculated by dividing the number of trips from each zone by the population in the zone.

In general, individual travel cost models are preferred to the zonal variant. This is because the zonal model is statistically inefficient, since it aggregates data from a large number of individual observations into a few zonal observations. In addition, the zonal model treats all individuals from within a zone as having the same travel costs, when this is often clearly not the case.

For both variants of the simple model, the demand curves are estimated by regressing visitation rates on socio-economic characteristics (for example income), estimates of the costs of visiting a site, and some indicator of site quality.

The simple travel cost models are estimated with data collected from surveys of either visitors to a recreational site, or households in a general population. In either case, the data requirements of this approach are potentially significant, since data are required on each individual's own characteristics (or population characteristics), as well as on the nature of their trip, the distance and time travelled, the costs of travel, and so on. These data are generally gained from some existing or specially commissioned survey. Site quality is often a somewhat intangible variable. Simple measures may concentrate on indicators such as angling catch rates, while others may be based on, for instance, biochemical indicators such as dissolved oxygen. The issue here, of course, is whether the measures selected coincide in any robust way with those, which individuals perceive to be relevant.

Unless the recreational site being valued is unique, most individuals will have access to a range of substitute sites that they could use for the same or similar recreational activity. Omitting these substitute sites will lead to bias in our benefit estimates, although there is no simple way of incorporating substitutes into this version of the model. Including all possible substitutes is obviously impractical, so judgement is needed on the part of the researcher. Multi-site models vary in their complexity and their ability to explain substitution behaviour. Often, restrictions are placed on site characteristics (for example models refer to some 'typical' site) or demand equations (for example 'pooled' models). Morey's (1984, 1985) 'share' model considers the allocation of an individual's fixed time budget between sites, thus being able to account for site substitution at the expense of being unable to explain the total amount of time allocated to recreation.

The general travel cost method is a technically well-developed valuation approach, which has been extensively employed over the past two decades. It is useful because it is grounded, at least in theory, on actual observed behaviour. However, the technical and data requirements should not be underestimated; travel cost is unlikely to be a low cost approach to non-market valuation.

Averting Behaviour

Perfect substitutability is the basis of the 'averting behaviour' technique, which looks at how averting inputs substitute for changes in environmental goods. For instance, expenditures on sound insulation can indicate householders' valuations of noise reduction; and expenditure on liming might reflect the benefits of reduced water acidification. The approach requires data on the environmental change and its associated substitution effects. Fairly crude approximations can be found simply by looking directly at the change in expenditure on the substitute good that arises as a result of some environmental change. Alternatively, the value per unit change of the environmental good can be determined. This is done by finding the marginal rate of substitution between the environmental commodity and the substitute private good from known or observed technical consumption data. The marginal rate of substitution is then multiplied by the price of the substitute private good, giving the value per unit change of the environmental good.

If the observed averting behaviour is not between two perfect substitutes, the benefits of the environmental good will be underestimated. For example, if there is an increase in environmental quality, the benefit of this change is given by the reduction in spending on the substitute market good required to keep the individual at their original level of welfare. However when the quality change takes place the individual will not reduce spending so as to stay at the original welfare level. Income effects will cause expenditure to be reallocated among all goods with a positive income elasticity of demand and so the reduction in spending on the substitute for environmental quality will not capture all of the benefits of the increase in quality.

Further problems with the approach include the fact that individuals may undertake more than one form of averting behaviour to any one environmental change, and that the averting behaviour may also have other beneficial effects which are not considered explicitly, for example sound insulation may also reduce heat loss from a home. Furthermore, averting behaviour is often not a continuous decision but a discrete one – a smoke alarm is either purchased or not, for instance. In this case the technique will again give an underestimate of benefits unless discrete choice models for averting behaviour are used.

Hence, simple avertive behaviour models can give incorrect estimates if they fail to incorporate the technical and behavioural alternatives to individuals' responses to quality changes. Nevertheless, although the technique has rarely been used, it is a potentially important source of valuation estimates since it gives theoretically correct estimates which are gained from actual expenditures and which thus have high criterion validity.

Hedonic Pricing Approach

Like avertive behaviour and travel cost, hedonic pricing also requires the weak complementarity assumption. However, it differs in that it operates through changes in the prices rather than quantities of private goods. A private good to which environmental quality is complementary can be viewed as a bundle of characteristics, which includes environmental quality. Then, individuals express their preferences for environmental quality by their selection of a particular bundle of characteristics. These preferences will be reflected in the differential prices paid for private goods in the market. The hedonic pricing approach applies econometric techniques to data on private good characteristics and prices to derive estimates of the implicit prices for environmental quality.

For instance, the location of residential property can affect the environmental attributes of that property, and potentially, therefore, the stream of benefits associated with residence. Let us assume that the expected stream of these benefits is capitalised in property values. Then the value of two properties which differ only in, say, the local air quality, will differ to the extent that people find one level of air quality preferable to the other. The difference in value can be viewed as the implicit price of the difference in air quality. Even when properties differ in many ways, not just in environmental quality, we might still be able to uncover the implicit prices if our data and statistical techniques are good enough.

The first stage of the hedonic price approach is to specify and estimate an hedonic price function, which relates house prices to all the relevant characteristics of the housing stock. More formally, it describes all points of equilibrium between sellers' offers of different environmental quality at different prices, and buyers' bids for environmental quality at those prices.

The marginal implicit price of each characteristic is given by the respective partial differential of the hedonic price function, that is the responsiveness of property price to a change in the characteristic in question. This price does not have to be constant, and indeed, we might expect the implicit price of a characteristic to fall as the quantity supplied increases. The price of one characteristic might also depend on the level of a different characteristic. However, rather than imposing any restrictions on the analysis, it is customary to see what functional form of the hedonic price function (and hence the form of the marginal implicit price function) fits the data best.

The preferred data include the sale prices associated with actual property market transactions; housing characteristics (for example number of rooms, type of neighbourhood); and environmental characteristics (for example noise, air quality). This presumed superiority of characteristics over, say, individuals' own valuations of their property does imply some stringent assumptions – in particular, that the housing market is in equilibrium. Violation of this assumption could result in a benefit estimate, which is either an upper or lower bound on the true benefit, although we should be able to tell which is the case in any particular study. Moreover, there is the obvious problem of how to measure qualitative variables such as air pollution in objective terms. Here, we encounter again the usual question of whether individuals' perceptions of qualitative variables can be related to objective, or policy-relevant measures. Similarly, can the changes in individual environmental variables be discerned, or must we work in terms of some 'overall' environmental quality? Furthermore, when variables are correlated, we face the uncertain trade-off of bias being introduced due to significant explanatory variables being omitted to reduce problems of multicollinearity.

We cannot generally use the estimated hedonic price function directly to value changes in environmental quality. This is because this function is a locus of supply and demand equilibria, whereas it is the demand (or bid) function alone which is relevant for benefit estimation. The hedonic price function will trace out the desired bid function only when all individuals are identical in tastes and preferences. However, even if all individuals are identical, the bid function cannot be identified if all individuals face the same implicit price for the environmental attribute to be valued. But this implicit price does at least reflect the marginal WTP of each individual for improved quality, and so could be used to value marginal changes in environmental quality. Unfortunately, environmental changes of most policy interest are often not marginal.

Hence, the second stage of the hedonic pricing approach generally involves regressing our implicit price estimates from the hedonic price function on physical and socio-economic variables thought to influence housing demand. If we assume that the supply of housing is fixed in the short run, this should allow us to identify the bid function we seek. However, the problem of identification should not be underestimated. All consumers within a housing market face the same equilibrium price schedule, or hedonic price function. Hence, the observation of a single consumer's behaviour provides only one point on that consumer's bid function. Other marginal prices are observed only for other individuals, so they provide no indication of the likely reaction of the original consumer to varying prices.

A number of solutions to the identification problem have been proposed. One option is to restrict variables or functional forms to be different in the second stage from those employed in the first stage, although this restriction cannot always be justified. The preferred alternative is to use data from spatially or temporally-separated markets, so that individuals do not face the same hedonic price function. This does, however, require that consumers are similar between these separate markets.

The hedonic approach relies on the assumptions of a freely functioning and efficient property market. Individuals have perfect information and mobility so that they can buy the exact property and associated characteristics that they desire and so reveal their demand for environmental quality. In reality the housing market is unlikely to be thus. A large part of the housing stock may be in the public sector and so allocated subject to price controls. Furthermore market segmentation may exist so that mobility between housing areas is restricted. Nevertheless, the hedonic approach is founded upon a sound theoretical base and is capable of producing valid estimates of benefits so long as individuals can perceive environmental changes. It has been employed to produce reliable estimates of the value of actual environmental changes.

Dose-Response Technique

The dose–response technique aims to establish a relationship between environmental impacts (the response) and some cause of the impact such as pollution (the dose). When the impact shows up in changes in the quantity or price of marketed inputs or outputs, the value of the change can be measured by changes in the total of 'consumer plus producer surplus'.

The technique is used extensively where dose–response relationships between some cause of damage such as pollution, and output/impacts are known. For example, it has been used to look at the effect of pollution on health, physical effects on materials and buildings, aquatic ecosystems, vegetation and soil erosion. The physical dose–response function is multiplied by a unit 'price' or value per unit of physical damage to give a 'monetary damage function'.

When the response to a 'dose' of pollution is marginal, we might be able to value the impact directly at current market prices. However, when output responses are not marginal, a more general approach is warranted. For instance, even if all prices remain unchanged, producers might change the quantity of other inputs into their production processes, implying different costs and a change in producer surplus. If the output price does change, consumers' consumption patterns will change, as will their consumer surplus. Hence, non-marginal output responses are likely to require a modelling approach to predict changes in prices and behaviour on both the supply and demand sides of the relevant markets. The effects of government intervention and market imperfection may also have to be incorporated (for example through shadow pricing).

Such prediction of market responses is complicated. Individuals will often make complex changes in their behaviour to protect themselves against any effects (averting behaviour). Farmers might switch to crop varieties which are resistant to air pollution. Materials corrosion might be countered by painting, switching to corrosion-resistant substitutes, or simply replacing the materials more often. A large number of markets might be involved, and modelling such an interrelated system can be an extremely sophisticated or fairly simple activity. The simpler models can provide useful estimates provided their shortcomings are recognised.

The specification of the dose–response relationship is crucial to the accuracy of the approach. The pollutant responsible for the damage needs to be identified as well as all possible variables affected. Large quantities of data may thus be required. Often there may be subtle but potentially significant forms of damage that are easy to observe and measure, for example leaf-drop and discolouring in vegetation. These can be used as variables in the dose–response function. However, some effects such as reduced plant vigour and less resilience to pests may be difficult to observe and measure directly. In such cases it is necessary to use an 'instrumental variable' in the dose–response function, that is one that is easily measurable and which indicates the level of damage, even if that damage cannot be measured directly. In some cases, the damage might not be economically relevant if individuals are not concerned by it. For instance, leaf loss and discoloration may have no impact upon the amenity or commercial value of forests.

To conclude, the dose–response approach is a technique that can be used in cases where the physical and ecological relationships between pollution and output or impact are known. The approach cannot estimate non-use values. The approach is theoretically sound, with any uncertainty residing mainly in specifying the dose–response relationships themselves, and in predicting behavioural responses to impacts. The approach may be costly to undertake if large databases need to be manipulated for physical and economic modelling. However if the dose–response functions already exist and impacts are marginal, the method can be very inexpensive, with low time demands, providing reasonable first approximations of the true economic value measures. In the case of air pollution damage, the dose–response function is in fact the main technique used to derive economic values.

Replacement Cost Method

This technique looks at the cost of replacing or restoring a damaged asset to its original state and uses this cost as a measure of the benefit of restoration. The approach is widely used because it is often easy to find estimates of such costs.

The approach is valid when it is possible to argue that the remedial work must take place because of some other constraint such as an environmental standard. Replacement will only be economically efficient, however, if the environmental standard was itself economically determined. Otherwise, the approach estimates only the costs of replacement: it is not a technique for benefit estimation.

Information on replacement costs can be obtained from direct observation of actual spending on restoring damaged assets or from engineering estimates of restoration costs. The technique implies various assumptions, for instance, that complete replacement is, in fact, feasible. In general, because of the highlighted potential for confusion between costs and benefits, the replacement cost technique should be used with some care.

Market Valuation

This attempts to estimate demand, supply, or production relationships from available price and quantity information, such that it is then possible to estimate and use market values to derive measures of value. From estimated demand and supply functions, estimates of consumer and producer surpluses can be calculated. This technique has mainly been used to estimate the demand for drinking water by municipal users (Gibbons, 1986). When water is used as an input for industrial purposes, production functions can be used. Theses use data on input use, capacity, and output to calculate the marginal productivity of each input, including water. This then allows the marginal value of water to be calculated by multiplying the marginal product of water by the output price to yield the value marginal product. At an efficient level of production the marginal products must be equal across inputs.

Simulation Models

Simulation models include bio-economic fishery models, crop yield models, and biological growth models. Here the biological or physical response to pollution or other physical event, and the resulting effect on human behaviour, such as increased fishing effort in response to changes in a population of fish are determined. It is then possible to incorporate the results from these models into economic models in order to obtain WTP measures associated with changes in exogenous factors.

Optimization Models

This type of technique is often used in economics to value water. There are a number of such optimisation models used to value water including mathematical programming and dynamic optimisation models. The models are based on a maximisation/minimisation subject to a constrained choice in order to find a “best” solution to a problem. Mathematical programming models tend to be static, or one-period, models, while dynamic optimisation models consider optimal choices for each period in a many-period framework.

Mathematical Programming Models - mathematical programming methods are often used to determine the value of irrigated water and groundwater when there is detailed data for a few representative agents. The models are used to model economic problems in which the economic agent (consumers, central planner, or firm) seeks to optimise (maximise or minimise) their welfare (e.g., surplus, costs, profit, or revenues) while facing inequality constraints that restrict their ability to choose certain levels of inputs or outputs. The models can be used to determine both marginal and non-marginal values for water as an input. Water enters mathematical programming models as an input constraint, such that its marginal value is found by relaxing the water constraint by adding a unit of water available for production and calculating the difference between the optimal value before and after relaxing the constraint. This marginal value of water is also known as the "shadow value" of water. Non-marginal changes can be evaluated similarly, whilst changes in the shadow value of water can be calculated for exogenous changes in output prices, input prices, or constraints.

Dynamic Optimization Models – These models rather than giving an optimal solution that maximises a single objective function for a specific period of time (as static models do), instead yields optimal paths of choice variables over multiple time periods. These models can again be used to determine both marginal and non-marginal values for water. The welfare measurement of changes in the quality or quantity of water is measured in a similar way as for the programming models. Such models have been used to measure the value of water allocation schemes, irrigation policies, and water quality projects, among other things.

Other Methods

The methods described below include three cost-based valuation techniques (cost savings methods, residual imputation methods, and income multiplier methods), an energy analysis approach, and an agricultural yield comparison approach.

Cost Savings Methods – This determines the value of water by the savings incurred by using the water in its current use versus the next best (cheapest) alternative. The method is fairly commonly used to value water for use in transportation, and can be applied to other uses of water as well. To value water as a means for transporting commerce, the cost savings resulting from not having to transport the same commerce via an alternative form of transit, typically by train, is a measure of the value of water for this use. As such the value of water for transporting goods is equal to the difference between the cost of transporting goods by train and the cost of transporting goods by boat. However, the approach does not adjust for the fact that time costs vary greatly between different transport modes (Gibbons, 1986). The method has also been used to value hydropower by estimating the difference between the cost of providing power using hydropower methods and the next cheapest power alternative. The approach equates cost savings with value, and so can be criticised on the ground that it implicitly assumed demand will be unresponsive to the change in costs.

Residual Imputation Methods – This is a form of a budget analysis technique that seeks to find the maximum return attributable to water or its use by calculating the total returns (profit, surplus, etc.) and subtracting off all non-water related expenses. The residual value is assumed to be the returns to water and represents the maximum amount the firm or farm would be willing to pay for water and still cover input costs (Naeser and Bennett, 1998). A net WTP amount can be calculated by subtracting water procurement costs from the residual value, which can be compared to values for water in other uses (Gibbons, 1986). This approach is sometimes categorised as a farm crop budget technique in applications to agriculture. A difficulty is that the residual return after subtracting the costs of all measured non-water inputs is the return to water plus all unmeasured inputs.

Income Multiplier Method – this technique employs the use of income multipliers. Income multipliers measure the circulation of expenditures through an isolated economy, tracing the flow of money through individual sectors of the economy. The total effect of expenditures in one sector of the economy on the total economy is thus measured by multiplying the income multipliers by the expenditures. However, changes in expenditures measure economic impact, not net economic value or WTP (Sorg and Loomis, 1984). As such they represent a redistribution of economic activity, which are transfers of surplus between regions, between people, and between industries that sum to zero in full employment economies. Changes in expenditures provide some evidence that the activity or amenity for which expenditures were made is valuable but do not provide guidance on the magnitude of this worth.

Energy Analysis - This method has been used to value ecosystems (Farber and Costanza, 1987) by examining the total biological productivity of an ecosystem to calculate a rough approximation of its value. The total energy captured by ecosystems is used as an estimate of the total potential the ecosystems have to do useful work for the economy. The approach is thus considered to provide an upper bound on the economic value of the ecosystem's products because not all products will be used by the economy. A dollar to energy conversion factor, measured in dollars per unit of energy, is used to derive the total economic value of the system for providing biological products for the economy by multiplying the energy potential by the conversion factor. The resulting welfare measure thus represents the total consumptive value of the ecosystem being valued. This suggests that non-consumptive benefits, such as some recreation values, are not accounted for in energy analysis-derived values. A difficulty with this method is that there is no reason to expect the conversion factor between energy and dollars to be even approximately constant.

Yield comparison method – This estimates the value of irrigated water as the observed difference in per acre returns between irrigated and non-irrigated land as estimated from farm budget information. This procedure forms the basis for the yield comparison method used to value irrigation water in agriculture. The method presumes the difference in net income associated with adding water to the production process is the producer's WTP for water. Thus, the difference in returns represents the maximum amount the producer would be willing to pay for the ability to use irrigation water. The approach does not require detailed information on crop mixes. However, the heterogeneity of land and crops planted, crop values, and other differences between irrigated and non-irrigated land question the validity of the difference in net returns being characterised as the net WTP for irrigated water.

Benefits transfer

It is not always possible for valuation studies to be conducted for all the benefits accruing to wetland values. To get around this benefits transfer is used; in valuation, it is inevitable that not all data will be readily available and budgetary constraints are likely to prohibit extensive collection of primary data. Thus, if a similar project has previously been undertaken elsewhere, estimates of its economic consequences might be usable as an indicator of the appropriate values for the impacts of the new project.

Such an approach has been termed ‘benefits transfer’ because the estimates of economic benefits are ‘transferred’ from a site where a study has already been completed to a site of policy interest. Using a benefits transfer approach appropriately will yield significant time and cost savings as compared to the time and resource intensive process of designing, testing and implementing a new valuation study. The benefits transferred from the study site could have been measured using any of the direct or indirect valuation techniques that are outlined in the later section on valuation techniques.

Environmental value transfer is commonly defined as the transposition of monetary environmental values estimated at one site (study site) through market based or non-market based economic valuation techniques to another site (policy site). The most important reason for using previous research results in new policy contexts is cost-effectiveness. Applying previous research findings to similar decision situations is a very attractive alternative to expensive and time consuming original research to quickly inform decision making. However, this technique of ‘benefits transfer’ is, fraught with difficulties and subject to a number of caveats, where any results and recommendations that transpire should explicitly be made conditional on these limitations.

If a benefits transfer approach was chosen to conduct an economic valuation then the main challenge faced is in finding the most appropriate studies to use in the transfer exercise. Criteria for transferring benefits between sites are suggested by Boyle and Bergstrom (1992) as:

- it should be the same goods or services that are being valued;
- relevant populations need to be very similar;
- the assignment of property rights concerning the wetland function under consideration should be the same.

The criteria for selecting studies for environmental value transfer suggested in the literature focus on where the environmental goods are found, the stakeholders and the study quality (Desvousges *et al.* 1992). Meta-analysis enables researchers to identify criteria for valid environmental value transfer or to test the convergent validity of value estimates (for further details on meta-analysis see Brouwer *et al.*, 1999). In the first case the data set is entirely used to determine the factors which help to significantly explain variances in valuation outcomes. In the second case the data set can be split for example in 2 parts, one of which is used for the first purpose and another to test whether the value estimates based on the significant factors fall within the confidence interval of the other half's estimates.

As more information about factors influencing environmental valuation outcomes becomes available, for instance through the meta-analysis, transfers across populations and sites seems to become more practicable, using either existing (secondary) information only or supplementing this information with new original (primary) data. However, very little published evidence exists of studies that test the validity of environmental value transfer. Moreover, in the few studies that have been carried out, the transfer errors are substantial (Brouwer, 1999).

Problems common to all methods of benefits transfer in addition to the requirement for good quality studies of similar situations, are the considerable potential for changes in characteristics between different time periods and the inability to value novel changes. As Green *et al.* (1994) point out, the quality of a cost-benefit analysis carried out using transferred benefits estimates will be no better than the quality of the transferred data itself, in the context of the study area to which it is applied. And Garrod and Willis (1994, p.23) suggest that, for the UK at least, even careful modification of available benefits estimates would not "yield transfer estimates which were reliable and robust enough to be used with confidence in policy applications." Benefits transfer might be more robust if essential scientific variables at different sites, based on ecosystem characteristics and processes, as well as socio-economic variables are considered.

Before a benefits transfer can be undertaken to attribute value to the change in the good or service provided by a wetland ecosystem, based on the results of another study, a step-by-step approach must be followed to ensure defensible results.

1. Identification of the impacts of a change in wetland ecosystem

2. Quantification of the Impacts

3. Selection of an appropriate study from which to transfer values

4. Transferral of values

In addition to identifying the good to be valued, it is also necessary to assess the current economic, political and cultural climate within which the study is being carried out, as these will influence the viability of the benefits transfer. The pattern of resource ownership might also determine the extent to which market prices exist for goods and services, and indicate the importance of wider social goals other than economic efficiency.

In order to attribute values to the impacts of a change in the wetland ecosystem, it is necessary to assess their significance. This requires determining the scale of the impacts in terms of area and/or numbers of people affected.

Choosing an appropriate set of studies from which to transfer values, involves matching the context of the previous economic study(ies), termed study sites, with the context of the current program or policy, termed the policy site. It is first necessary to define the environmental good or service, or human health effect which is to be valued, followed by assessing the suitability of the candidate studies based on similarities between the policy site and the study sites. Using the categories of transfer criteria outlined in the environmental economics literature, helps to ensure that the selection of candidate studies leads to defensible transfers. Suitability is determined based on similarities between the policy site and the study sites in the following areas:

- Geographic location
- Population
- Environment
- Timeliness of Data
- Economic Measure
- Estimated Values

This selection process must be done on a case-by-case basis – there is no "one-size fits all" solution. The specific characteristics of both the policy and study must be comparable.

Once the impacts have been identified and quantified it is then possible to undertake monetary valuation of the impacts, the aggregation of values over the relevant population – the multiplication of the estimated value from an appropriate study by the number of people affected. For this to take place, the values must be standardised in terms of both a base year AND currency involved. This will give an indication of monetary benefits of the change in the wetland ecosystem.

Summary of Valuation Techniques and Their Relative Strengths

To conclude this section on we set out below brief summaries of some of the main techniques and their relative strengths and weaknesses.

Contingent Valuation Method (CVM):

Range of Applicability	Extensive, since it can be used to derive values for almost any environmental change. This explains its attractiveness to 'valuers'. Only method for eliciting non-use values. Successfully applied in developing countries to water supply, water quality, forest access.
Procedure	Involves administering a carefully worded questionnaire which asks people their WTP and/or WTA compensation for a specified environmental change. Econometric analysis of survey results is generally required to derive mean values of WTP bids and to estimate the determinants of respondents' WTP. Literature tends to suggest that most sensible results come from cases where respondents are familiar with the asset being 'valued'.
Validity	The literature has identified various forms of potential bias. 'Strategic bias' arises if respondents intentionally give responses that do not reflect their 'true' values. They may do this if they think there is potential to 'free ride'. However, there is limited evidence of strategic bias. 'Hypothetical bias' arises because respondents are not making 'real' transactions. Costs of studies usually limits the number of experiments involving real money (criterion validity), but some studies exist. Convergent validity is good. Construct validity – relating value estimates to expectations of values estimated using other measures – is debated, especially the marked divergence in many studies between WTP and WTA compensation.
Reference	Case material is extensively reviewed in Mitchell, R. and Carson, R. (1989).

Contingent Ranking Method (CRM):

Range of Applicability	Extensive. Limited number of studies exist and are confined to 'private goods' – that is goods purchased in the market place. It is unclear how extensive the range of application could be for environmental goods but this is under investigation in the context of house location decisions.
Procedure	Individuals are asked to rank several alternatives rather than express a WTP. Alternatives tend to differ according to some risk characteristic and price. Method could be extended to a ranking of house characteristics with some 'anchor' such as the house price being used to convert rankings into WTP.
Validity	Not widely discussed in the literature but is theoretically valid. Too few studies exist to test other validity measures but initial results suggest CRM WTP exceeds CVM WTP.
Reference	Magat, W., Viscusi, W.K. and Huber, J. (1987).

Conventional market approaches (including dose–response, replacement cost, and opportunity cost approaches):

Range of Applicability	Extensively used where ‘dose–response’ relationships between pollution and output or impact are known. Examples include crop and forest damage from air pollution, materials damage, health impacts of pollution, output losses from soil erosion, sedimentation from soil erosion. Limited to cases where there are markets or where shadow prices can be estimated – that is the method cannot be used to estimate non-use values. Replacement cost approaches also widely used because it is often relatively easy to find estimates of such costs. Replacement cost approaches should be confined to situations where the cost relates to achieving some agreed environmental standard, or where there is an overall constraint requiring that a certain level of environmental quality is achieved. Opportunity cost approaches are very useful where a policy precludes access to an area – for example estimating forgone money and in-kind incomes from establishment of a protected area. Numerous applications in developing countries.
Procedure	Dose–response: takes physical and ecological links between pollution (‘dose’) and impact (‘response’) and values the final impact at a market or shadow price. Most of the effort usually resides in the non-economic exercise of establishing the dose–response links. Multiple regression techniques often used for this. Replacement Cost: ascertain environmental damage and then estimate cost of restoring environment to its original state. Opportunity Cost: ascertain functions of displaced land use and estimate in-kind and money incomes from those uses. May require detailed household surveys to establish economic and leisure activities in the area in question.
Validity	Dose–response: theoretically a sound approach. Uncertainty resides mainly in the errors in the dose–response relationship for example where, if they exist, are threshold levels before damage occurs? Are there ‘jumps’ (discontinuities) in the dose–damage relationship? An adequate ‘pool’ of studies may not be available for cross-reference. Criterion validity not relevant since presence of ‘real’ markets tends to be a test in itself – that is revealed preferences in the market place are being used as the appropriate measure of value. Replacement Cost: validity limited to contexts where agreed standards must be met. Opportunity Cost: sound measure of damage done by a given land use that precludes other activity. More sophisticated estimates would include lost consumer surplus.
Expense	Dose–response can be costly if large databases need to be assembled and manipulated in order to establish dose–response relationships. If dose–response functions already exist, the method can be very inexpensive and quick. Replacement cost is inexpensive if engineering data exists.
Reference	US Environmental Protection Agency (1985).

Avertive Behaviour:

Range of Applicability	Limited to cases where households spend money to offset environmental hazards, but these can be important – for example noise insulation expenditures; risk-reducing expenditures such as smoke-detectors, safety belts, water filters, and so on Application in developing countries uncertain – probably small. Has not been used to estimate non-use values though arguable that payments to some wildlife societies could be interpreted as insurance payments for conservation.
Procedure	Whilst used comparatively rarely, the approach is potentially important. Expenditures undertaken by households and designed to offset an environmental risk need to be identified. Examples include noise abatement, reactions to radon gas exposure – for example purchase of monitoring equipment, visits to medics, and so on
Validity	Theoretically correct. Insufficient studies to comment on convergent validity. Uses actual expenditures so criterion validity is generally met.
Expense	Econometric analysis on panel and survey data is sometimes needed. Can be fairly expensive.
Reference	Dickie, D., Gerking, S. and Agee, M. (1991).

Travel Cost Method:

Range of Applicability	Generally limited to site characteristics and to valuation of time. Former tends to be recreational sites. Latter often known as discrete choice – for example implicit value of time can be estimated by observing how choice between travel modes is made or how choice of good relates to travel time avoided (last case has been used to value women's water collection time in developing countries). Cannot be used to estimate non-use values.
Procedure	Detailed sample survey needed of travellers or households, together with their costs of travel to the site. Complications include other possible benefits of the travelling, and presence of competing sites.
Validity	Theoretically correct, but complicated when there are multi-purpose trips and competing sites. Some doubts about 'construct validity' in that number of trips should be inversely correlated with 'price' of trips – that is, distance travelled. Some UK studies do not show this relationship. Convergent validity generally good in US studies. Generally acceptable to official agencies and conservation groups.
Reference	Willis, K. and Benson, J. (1988).

Hedonic Property Pricing:

Range of Applicability	Applicable only to environmental attributes likely to be capitalised into the price of housing and/or land. Most relevant to noise and air pollution and neighbourhood amenity. Does not measure non-use value and is confined to cases where property owners are aware of environmental variables and act because of them (as with avertive behaviour).
Procedure	Approach generally involves assembly of cross-sectional data on house sales or house price estimates by estate agents, together with data on factors likely to influence these prices. Multiple regression techniques are then needed to obtain the first estimate of an 'implicit price'. A further stage of analysis is required since the multiple regression approach does not identify the demand curve directly.
Validity	Theoretically sound, though market failures may mean that prices are distorted, that is markets may not behave as required by the approach. Data on prices and factors determining prices often difficult to come by. Limited tests of convergent validity but generally encouraging results.
Reference	Brookshire, D. *et al.* (1982).

Hedonic Wage-Risk Estimation:

Range of Applicability	Limited to valuation of morbidity and mortality risks in occupations. Resulting 'values of life' have been widely used and applied elsewhere, for example in the dose–response approach.
Procedure	As with other hedonic pricing methods, the approach uses multiple regression to relate wages/salaries to factors influencing them. Included in the determining factors is a measure of risk of accident. The resulting 'wage premium' can then be related to risk factors to derive the so-called value of a statistical life.
Validity	Theoretically sound. Convergent validity may be tested against CVM of risk reduction, but wage-risk approach measures WTA compensation not WTP.
Reference	Marin, A. and Psacharopoulos, G. (1982).

Annex E: Further Issues for Consideration when Valuing Wetland Ecosystem Functions in Practice

1. Spatial and Temporal Scale

The scale of evaluation is determined by the issue that is under investigation. For a specific isolated external impact, evaluation may be restricted to a limited number of affected variables. Where broader changes are involved (e.g. a change in land use in a catchment), partial analysis of a number of integrated parameters may be required. Because of the costs and effort that are involved, full valuations are usually avoided unless they are absolutely necessary e.g. a situation where an entire catchment is under threat.

The spatial scale (or accounting stance) of a study is determined by the extent of the population that is affected by the impact under investigation. The accounting stance should be as encompassing in this respect as possible. Where the impact incurs only changes in direct uses of a wetland, the affected population is existing and potential resource users. This population does not, however, necessarily live in close proximity to the resource as they may travel considerable distances to use it. Indirect use values may not be site-specific in terms of those who benefit, e.g. interception of flood waters by irrigation may yield benefits far downstream. Non-use benefits are derived over a wide geographical area, but are likely to be subject to 'distance decay' away from the site. In practice, a pragmatic accounting stance has to be adopted in specifying the scale, where the gains in accuracy are balanced against the costs of spreading the scale wider.

The temporal scale, combined with the discount rate (discussed below), influences the present value of the streams of costs and benefits. The calculation of expected future costs and benefits involves estimating future demand. This is necessarily unknown but a range of possible values can be obtained through the assessment of likely scenarios and application of sensitivity analysis. The temporal scale also determines the trade-off between considering long run versus short run values. Decisions are more constrained and responses quite different in short run contexts. Most public policy contexts relate to the longer term, though there are some circumstances, such as drought planning, for which short run values are more appropriate.

2. Aggregation and double counting

This report advocates adoption of a functional approach to wetlands. This involves considering the goods and services provided by wetlands in relation to environmental structures and processes. It does, however, raise issues that require attention in the aggregation of data on the benefits provided:

- While the adoption of a functional perspective is advocated as the correct way to identify wetland goods and services, if each of them is identified separately, and then attributed to underlying functions, there is a likelihood that benefits will be double counted. Benefits might therefore have to be allocated explicitly between functions. For instance, Barbier (1994) noted that if the nutrient retention function is integral to the maintenance of biodiversity, then if both functions are valued separately and aggregated, this would double count the nutrient retention which is already 'captured' in the biodiversity value. Some functions might also be incompatible, such as water extraction and water recharge, so that combining these values would overestimate the feasible benefits to be derived from the wetland. Studies that attempt to value the wetland as a whole based on an aggregation of separate values tend to include a certain number of functions although these studies do not usually claim to encompass all possible benefits associated with the wetland.

- Some functions of wetlands may be mutually exclusive and, therefore, cannot be aggregated. For example, aggregation of the values for both extraction of water and recharge of water would overestimate the benefits that could feasibly be derived from a wetland.
- Interactions can occur between functions. For example, conservation goals may require alteration to the harvesting regime employed for reed beds, which reduces the gross margins of the beds. Some functions may be complementary; for instance, nutrient retention can promote biomass production.

In practice, the ability to use wetlands repeatedly or simultaneously for different uses means that competition and complementarity are important considerations in valuing wetland ecosystems. Wetland management would ideally be considered under a general equilibrium framework, though this is in practice extremely difficult. This also means that total valuation (estimation of the full value of a wetland ecosytem) is undertaken only when necessary. Management decisions are more commonly assessed using impact analysis (which assess the damage arising only from a particular impact) or partial valuation, based on a sectoral approach or on specific functions of a wetland. Such a partial approach means that a number of considerations must be taken into account. Firstly the different ways of calculating values may result in fundamentally different definitions of value, for example, which are specific to certain time frames that differ between the uses considered. Secondly, values may be based on average or marginal concepts, which are quite different concepts. Use of marginal values is required for the purposes of efficient allocation.

3. Allocation over time

It is frequently necessary to choose between options that differ in temporal patterns of costs and benefits, or that differ in their duration. Discounting provides a common matrix that enables comparison of costs and benefits that occur at different points in time. Use of discounting is integral to cost benefit analysis and cost effectiveness analysis.

Discounting converts the stream of costs and benefits over time into a stream of 'present' values. The difference between the value of the discounted benefits and costs is referred to as the 'net present value' (NPV). A management or policy option is economically viable only if NPV is positive, as described in Equation 1:

$$NPV = \sum_t \frac{(B_t - C_t)}{(1 + r)^t} > 0 \qquad (1)$$

where B_t and C_t are benefits and costs in year t respectively, and r is the discount rate.

The rationale for discounting is that costs and benefits that occur in the future are not valued as highly as those that occur in the present. There are two explanations for this:

(1) Time preference (or the 'consumption rate of interest'). Individuals prefer consumption in the present over consumption in the future. Reasons for this include:

the risks involved in delayed consumption;

anticipation of increased wealth in the future, which reduces the relative worth of postponed consumption (i.e. decreasing marginal utility of consumption);

'pure' time preference or myopia.

(2) The opportunity cost of capital. Financial capital that is not consumed in the present can be invested and expected to increase in value by the rate of interest. There is, therefore, an opportunity cost associated with present consumption of financial capital, which is the return that could be derived from its investment (as indicated by the rate of interest).

The choice of discount rate can have a significant effect on economic viability of management options and their relative economic ranking. It effectively signals the rate at which future consumption is to be traded against consumption in the present. Use of a high discount rate discriminates against the future. It discriminates against options that involve high initial costs and a stream of benefits that extends far into the future (e.g. creation or restoration of a wetland). Instead, it favours options that have immediate benefits and a lag in incurring costs. This has been described as the 'tyranny' of discounting (Pearce et al 1989).

High discount rates tend to be justified based on the opportunity cost of capital, though to be correct this is relevant only for financial analysis, which is not the examined here. In general, they are likely to encourage depletion of non-renewable natural resources and exploitation of renewable natural resources, reducing the inheritance of natural capital for future generations. Low discount rates favour the future but could discriminate against and hamper immediate economic development. They encourage investments which would otherwise have not been viable and which could be associated with an even more rapid depletion of natural resources (Fisher & Krutilla, 1975). The impact that the discount rate has on the environment is therefore ambiguous, and it is not clear that the call for use of lower discount rates to incorporate environmental concerns is generally valid.

A social rate of discount is used to evaluate the impact of management options on intergenerational welfare. Such evaluations take intergenerational welfare into consideration. The maintenance of future welfare can be regarded as a public good, in which private individuals will tend to under-invest. As a result, the social discount rate is lower than the equivalent rate of discount for individuals. The social discount rate is measured either as the social rate of time preference (SRTP) or the social opportunity cost of capital (SOC). Care has to be taken in developing country contexts, where the use of consumption rates of interest (which are likely to exceed four to six percent) may not adequately account for concerns about the inheritance of environmental problems by future generations.

The social discount rate can also be adjusted to reflect temporal trends in the net benefits of environmental preservation and development. The net benefits of such preservation are likely to increase over time as demand for environmental services rises under conditions of limited or declining supply. Conversely, the net benefits of development projects are expected to decline over time due to technological advancement. These trends can be incorporated into economic evaluation through appropriate adjustment of the social discount rate: by decreasing the discount rate applied to preservation benefits and increasing the rate applied to development benefits (Hanley & Craig 1991).

4. Risk and uncertainty

In the case of risk, meaningful probabilities can be assigned to the likely outcomes. In the case of uncertainty, probabilities are entirely unknown. Risk can be incorporated into an evaluation by attributing probabilities to possible outcomes, thereby estimating directly the expected value of future costs and benefits (Boadway and Bruce, 1984) or their 'certainty equivalents' (Markandya and Pearce, 1988). A premium for risk can be incorporated into the discount rate used for the analysis, but such adjustment is arbitrary, often subjective and attributes a strict (and unlikely) time profile to the treatment of risk and is not recommended for these reasons.

In an economic evaluation, uncertainty is associated with physical outcomes and their economic consequences. For wetlands, the necessary assessment of possible outcomes and the likelihood of perturbations to what is a highly complex system is inevitably fraught with difficulty. However, this is a necessary component of an economic evaluation. For each management or policy option under consideration, the range of possible impacts needs to be identified and quantified as far as possible. A particularly important issue relating to uncertainty in physical effects is the possible existence of thresholds beyond which disproportional and irreversible effects can occur. Such thresholds result in disproportional impacts and an inability to reverse consequences in the future (discussed further below).

There is also uncertainty that relates to the physical and economic conditions that will prevail in the future. For example, a change in regulations concerning agricultural production could cause farmers to respond with a change in land use, which could impact on nutrient concentrations in run-off and thereby affect the value of the nutrient retention function provided by a wetland. Likewise, individuals can alter their behaviour in response to changes in wetland functions. For example, an increase in flooding might be responded to by farmers through a change in cropping patterns. Such uncertainties can influence projected benefits and so also need to be incorporated into any evaluation of options.

Uncertainty is incorporated into economic evaluations through the use of sensitivity analysis or scenario analysis. In sensitivity analysis, various possible values are used for key variables in the evaluation such as the discount rate, the extent of functions, and economic values. This provides a range of estimates within which the true result can be expected to fall. It can create ambiguity but is a necessary component of any economic evaluation. Scenario analysis can also be used to incorporate uncertainty through comparison of results using parameter values that represent different possible future scenarios.

Costanza (1994, p.97) points out that "most important environmental problems suffer from true uncertainty, not merely risk." In an economic sense, such pure uncertainty can be considered as 'social uncertainty' or 'natural uncertainty' (Bishop, 1978). Social uncertainty derives from factors such as future incomes and technology, which influence whether or not a resource is regarded as valuable in the future. Natural uncertainty is associated with our imperfect knowledge of the environment and whether it has unknown features that may yet prove to be of value. This may be particularly relevant to ecosystems for which the multitude of functions that are performed have historically been unappreciated. A practical means of dealing with such complete uncertainty is to complement the use of a cost-benefit criterion based purely upon monetary valuation with a safe minimum standards (SMS) decision rule (Ciriacy-Wantrup, 1952; Bishop, 1978; Crowards, 1996), as discussed below.

5. Irreversible change

Irreversible impacts, for instance the extinction of species or exhaustion of minerals, are not accounted for in the standard procedures for economic evaluation. Under such circumstances, account needs to be taken of the uncertain future losses that might be associated with potential irreversible change. Some protection to the interests of future generations can be offered through the imposition of the safe minimum standards decision rule (Ciriacy-Wantrup, 1952; Bishop, 1978; Crowards, 1996).

The safe minimum standards decision rule recommends that when a development activity that impacts on the environment threatens to breach an irreversible threshold, that conservation is adopted unless the costs of foregoing the development are regarded as 'unacceptably large'. It is based on a modified principle of minimising the maximum possible loss and therefore differs from routine trade-offs which are based on maximising expected gains e.g. cost-benefit and risk analysis. However, activities that result in potential irreversible change are not rejected if the associated costs are regarded as intolerably high.

A critical aspect in the application of the safe minimum standards decision rule is specification of the threshold for unacceptable costs of foregoing development. The degree of sacrifice is determined through full cost-benefit assessment of the development option, including estimable costs of damage to the environment. The decision as to whether conservation of natural resources can be justified (and rejection of the development activity) is political, constrained by society's various goals. In this sense, safe minimum standards provides a mechanism for incorporating the precautionary principle into decision-making: society may choose to conserve even in the absence of proof that damage will occur in order to limit potential costs in the future (Crowards, 1997).

The concept of safe minimum standards has usually been applied to endangered species. However, it could equally be applied to irreversible impacts that threaten wetlands. Where thresholds of wetland processes are threatened with irreversible change, the use of safe minimum standards provides a decision framework that gives more weight to concerns of future generations. It promotes a more sustainable approach to current development and can provide an appropriate supplement to standard analysis of economic efficiency.

Safe minimum standards are closely related to sustainability considerations (Pearce and Turner, 1990). Sustainability essentially requires that the stock of natural capital available in the future is equivalent to that available at present. The concept of sustainability has been roughly partitioned into two approaches: weak sustainability and strong sustainability (Turner, 1993). Weak sustainability requires that the total stock of capital, whether man-made or natural, is maintained, and rests upon the assumption of substitutability between these two types of capital. Economic theory suggests that decreases in supplies of natural resources cause their prices to increase, which encourages more efficient use of natural resources, substitution with other goods, and technological advancement. However, complete substitution is not always be possible due to physical limits on the efficiency and availability of opportunities for substitution, the question of whether man-made capital can fully compensate for all the functions provided by complex ecosystems, and the existence of 'critical' natural capital and thresholds beyond which reversal is not possible. The more stringent interpretation of 'strong' sustainability requires that the total stock of natural capital is non-declining. Under this criterion, projects should either conserve the natural environment or ensure that losses incurred are replaced or fully compensated for in physical terms by the implementation of 'shadow projects' (Barbier et al, 1990).

An alternative mechanism that can be employed to account for potential irreversibility in the analysis of discrete development-conservation choices (e.g. if a development entails exploitation of a wetland to permanent loss) is to include the preservation benefits foregone as opportunity costs in the cost benefit analysis. Future development benefits that occur as a result of relative price effects and technology changes are discounted and also included in the analysis. This approach is known as the Krutilla-Fisher algorithm (Krutilla and Fisher, 1985). Irreversible change can also be incorporated into the evaluation through adjustment of the social discount rate to allow for temporal trends in the benefits of preservation (discussed earlier).

6. Data limitations

It is inevitable that some of the data required for an economic evaluation will not be readily available. Budgetary constraints often limit extensive collection of original data. Where data are limited, this should be acknowledged and the measures taken in response to this limitation clearly specified. The results and recommendations should be made explicitly conditional on these limitations. The various techniques used to value non-marketed goods and services are each associated with specific data limitations.

Annex F: Practical Valuation of Wetland Functions

This annex provides an overview of some of the more economically significant functions provided by wetlands (existing and potential newly created), along with a brief discussion of the application of possible techniques for their valuation. The functions are arranged according to whether they are hydrological, biogeochemical or ecological functions.

F1.Valuation of Hydrological Functions

F1.1 Flood water detention

"The short or long term detention and storage of waters from overbank flooding and/or slope runoff"

By diverting floodwaters from entering into rivers directly, for future more gradual release, wetlands reduce the peak river discharges and consequently reduce flood damage downstream. In evaluating the benefits of the flood water detention function, an assessment is required of impacts on the extent of flooding as a consequence of the reduction in peak flow. Both the potential benefits of flood water detention and flood control will be assessed in this section. In addition, flood water detention can be important in terms of wildlife habitat provision, and as a component of other functions such as sediment retention and nutrient retention/export.

The procedures involved in assessing the economic value of the flood control function of a wetland involve four distinct stages. These are:

I Assessing the potential for downstream flooding that will be influenced by the wetland (i.e. the assets at risk downstream).

II Determining the extent to which the wetland influences downstream flooding, and how flooding would be affected were the wetland to be removed (the with- and without- comparison).

III Identifying the potential for floods to damage resources and structures downstream.

IV Estimating the economic value of the wetland's flood control function.

Stages II and III are important because it is not the percentage of flood water that the wetland diverts, nor even the reduction in the physical extent of downstream flooding, that apportions economic value to the flood control function. It is the influence that this potential flooding will have on resources regarded by society as worth preserving that determines the value associated with the ability of a wetland to reduce flooding impacts.
In the final stage (IV), there is a choice of methods that can be employed to estimate economic value of flood control, and it will be up to the analyst to decide which method(s) to employ. While some are theoretically preferable to others, in that they

produce values based on the benefits that society derives from flood control, other 'second best' measures are often easier to determine in practice.

Assessing the potential for downstream flooding that will be influenced by the wetland.

A. Is the wetland performing a storm water storage function?

B. Is there a potential flooding problem downstream?

1. Is there a history of flooding in the catchment?

2. Is there evidence of flood management activities (past or present) in the catchment?

3. Is there significant human activity (e.g. buildings or farming and land drainage) adjacent to the river downstream?

4. Is the wetland's water storage capacity 'significant' (i.e. could it influence downstream flood potential) compared to the discharge of the river?

If the answer to 1. **or** 2. is yes, then proceed with evaluation.

If the answer to 3. **and** 4. is yes, then proceed with evaluation.

If otherwise, there may be insufficient economic value attributable to the wetland's water storage function to warrant a detailed evaluation.

Determining the extent to which the wetland influences downstream flooding, and how flooding would be affected were the wetland to be removed (the "with"- and "without"-comparison).

A. What is the likely influence of the wetland's water storage function on downstream flooding?

1. As with 4 above: how 'significant' is the wetland's storage capacity compared to discharge of the river?
2. How much water is the wetland likely to divert from the river's storm discharge?

   [Require information on storm discharge at the point of the wetland, to estimate probable future discharges, as well as the likely capacity of the wetland to reduce this discharge.]
3. How does the reduction in discharge correspond to flood levels downstream: e.g. does one m^3 of water storage correspond to one m^3 reduced flooding downstream?
4. What is the maximum additional water storage capacity of the wetland? Does this vary significantly according to factors such as time of year?

[There will be a limit to the additional water storage capacity; so that the significant reduction in downstream peak discharge may be restricted to smaller flood episodes: the reduction is unlikely to remain a fixed percentage of discharge.]

5. How does floodwater from the river in question synchronise with floodwaters from other tributaries to produce peak flood levels?

   [It is feasible that simply by delaying the discharge of water, flooding downstream could be made worse, depending upon the synchronisation of different tributaries.]

A number of these points, especially those relating to catchment -level issues are raised in Larson (1986). Interactions at the scale of the watershed are important, since “although it is possible for an isolated wetland to perform a significant flood control function, effective flood control is more often the result of the interrelationship of a series of wetlands within a particular watershed.” (Sather and Smith,1984, p.5).

B. What is the likely degree of downstream flooding if the wetland remains undisturbed?

   e.g. historical episodes; flooding predictions or records; efficacy of flood mitigation measures. Factors such as location, area, depth and duration (especially, more or less than 12 hours) will be important.

C. How frequently might such floods be expected? (return periods or probabilities)

   e.g. historical episodes; flooding record and predictions; efficacy of flood mitigation measures.

D. From A., B. and C. above, what degree of (increased) flooding might be expected if the wetland’s flood water storage function were negated?

E. Hence, to what (quantitative) extent is the wetland expected to influence downstream flooding (location, area, depth, timing and duration)?

Identifying the potential for flood damage downstream:

What are the land (or river) uses in potential flood areas?

If, for instance, flooding is most likely to affect forest areas, parkland or other wetlands downstream, the damage may be minimal and short term. However, if an urban area is under threat, damage costs could be considerable and longer term.

Velocity reduction and erosion control:

The flood control function can also have benefits in terms of control of bank erosion caused by peak river discharges. This is achieved through the retention and delayed gradual release of water. Some wetlands continually reduce the velocity of surface water flows, not only during high discharge episodes, further limiting erosion downstream. The value of erosion control is determined by the extent of potential erosion and its impact on social welfare. If erosion of a river bank would result in loss of marginal grazing land, for example, the value of its control is low, but if it were to undermine the foundations of a building, the value would be higher. The preceding sections on identifying the potential damages downstream are relevant here. The most likely valuation approach will be to assess the 'damage costs avoided' by maintaining the wetland, which is outlined in detail below.

Valuation techniques:

- *Hedonic pricing.* Hedonic pricing can be used to analyse the price differential for properties that are at risk from flooding. It entails analysis of all variables that could affect price, such as location, size, aspect, and age of property. Use of hedonic pricing requires existence of a property market and existence of known and distinct risks of flooding. It is a complicated procedure. Impacts of flooding on property prices can be countered by defensive expenditures to reduce flood damage (Holway and Burby, 1990). A further complication is that perceived risk of flooding and the resultant impact on house prices can diminish as memories of previous flooding episodes fade (Tunstall, Tapsell & Fordham, 1994). Also, house prices may reflect flood hazards only where flooding has occurred relatively recently, regardless of the expected frequency of flooding (Tobin & Newton, 1986).

- *Contingent valuation.* Flood water control can be valued by asking the affected population what they would be hypothetically willing to pay to either avoid flooding (of some area of interest) or to avoid an increase in the frequency of flood episodes. Given the analytical and resource demands of contingent valuation survey, this is best limited to valuation of the impacts of flooding that are not non-marketed, such as impacts on unique ecosystems.

- *Damage costs avoided.* The costs that would be incurred if flood control provision (e.g. the flood protection provided by a wetland) was not present are given by the damage costs. These can be divided into direct costs, indirect costs and intangible costs.

  The direct costs of flooding are incurred by physical contact with the floodwaters. Costs of damage to the built environment are determined by the type of building (e.g. residential, commercial, industrial) and factors such as the design, function, density and age of the buildings. Cost estimates can be obtained from relevant publications (e.g. the 'FLAIR blue book' used in the UK (N'Jai et al, 1990), government agencies, or site specific surveys conducted by government agencies or insurers. In determining the costs of damage to movable assets, account needs to be taken of avertive action. For example, Tunstall et al (1994) found in a study of flooding in Maidenhead in the UK that the reduction in damages due to avertive action was 'substantial'. Higgs

(1992) allows for a 5-10% reduction in damage costs due to items being moved away in advance from flood prone areas.

Flooding also imposes costs on productive activities in the non-built environment. Damage to natural ecosystems (wetlands, woodlands and meadows) may be minor and temporary. However, the costs can be substantial for intensive agriculture. Losses in returns to agricultural production are determined by the depth, extent and duration of flooding, the effluent and silt content of the flood waters, types of crop, expected yields and price. Silt exacerbates the volume of flooding and is itself a cause of damage; Clark (1985) estimated that silt accounted for 20% and 7 % of urban and rural flood damage respectively for a study in the US. Returns from agriculture, as the opportunity cost of wetland creservation, are calculated by Turner et al (1983) based on detailed analysis of output, fixed and variable costs and transfer payments (agricultural subsidies). Estimates of standard losses in agricultural gross margins due to flooding may also be available from official publications (e.g. the Farm Management Pocketbook for the UK). Long term impacts on agricultural production through continued exposure to inundation are reflected in the value of the land. Flooding affects the land use categorisation of land and this is reflected in the market price (Boddington, 1993); average price data for land use categories is often available from official publications.

Warning of impending floods allows people to take action to reduce potential damages. For instance, in a study of flooding in Maidenhead, UK, the reduction in damages due to such action were found to be ‘substantial’ (Tunstall et al. , 1994). In an assessment of potential flood damages, Higgs (1992) allows for reduced damage costs of 5% - 10% as a result of goods being moved away from flood-prone areas when a flood is expected.

Flooding also results in indirect and intangible costs. Indirect costs are caused by disruption to physical and economic linkages in the economy. They include costs of implementing immediate emergency measures; reduced production, and the knock-on effects of this on production elsewhere; impacts on transport; and increases in living expenses. Intangible costs by definition cannot be readily quantified. Examples include psychological effects (stress caused by flooding and worry about future events) and poor health caused by flooding. Some costs formerly described as intangibles are now being quantified such as the effects of disruption and evacuation (Green & Penning-Rowsell, 1986; 1989). Intangible costs could be more significant that the direct damages of a flood episode (Green & Penning-Rowsell, 1989). It is best to acknowledge that such costs are expected but cannot be valued and that the total cost of damage (and hence the value of the wetland flood protection function) is underestimated as a result.

- *Defensive expenditures.* Defensive expenditures provide only a minimum estimate of the benefits of flood water control as they may omit costs of flooding against which defensive actions are not taken. Furthermore, defensive expenditures tend to be low relative to potential damages as individuals underestimate the likelihood of flooding and over-estimate their ability to cope with its effects (Tunstall et al, 1994). Defensive expenditures include relocation of assets (buildings, nature reserves and livestock (Boddington, 1993), rewiring of electrical points above expected flood levels and raising of houses on stilts or piles (Tunstall et al., 1994). Relocation may not be to a site that is a direct substitute. Costs of relocation therefore need to be attributed accordingly between the various benefits, and any disadvantages also taken into account.

- *Replacement Cost.* The replacement cost of flood control can be determined, for example, through the use of shadow projects. In the case of the flood water control function of a wetland, a shadow project could entail creation or restoration of another wetland that would performed the same function within a given catchment. This would also replace other functions of the wetland and would, therefore, be particularly appropriate in a situation where total loss of a wetland is threatened. Locally relevant costings for wetland creation or restoration are likely to be sparse (though mitigation banking has led to considerable creation and restoration of wetlands in the U.S.). There is uncertainty associated with 'engineered' ecosystems and the functions that they can perform. This will be more pronounced the more geographically distant are the original and new locations, and economic values in terms of who can derive the benefits are also likely to alter with increasing distance. There is also the question of 'authenticity', where the natural or original version of a resource may be preferred to even an exact replica, thereby influencing the value (especially amenity and non-use values) attributed to the ecosystem. For instance, Kosz (1996) finds a strong preference amongst Austrians for wetlands in an unchanged state, even if artificial manipulation allows natural conditions to be simulated "perfectly". For shadow projects that entail a change in land use for a site (e.g. taking land out of agricultural production) the opportunity costs of this must also be included in the analysis.

- *Current Flood Protection Measures*
  Identifying flood protection measures already in place in the catchment can provide useful indicators as to the potential extent and possible costs associated with any increased flooding:

  - If flood protection measures are in place, then presumably there **is** considerable flooding potential; costs of improving facilities already in place may be substantially less than building entirely new facilities.

  - If there are no flood protection measures in place, then flooding might not be expected to be a major problem; even if it is, costs of implementing new flood protection schemes could exceed the benefits of reduced flooding.

  Artificial flood control measures:

  A. Given the expected degree of increased flooding, what preventive measures would be required?

Possible measures could involve construction of drainage channels, levees, barrages and flood walls, pumping and drainage systems, channel alteration, or dam construction; or improving such facilities already in place.

B. What are the estimated costs of such an enterprise?

This information may be available from local water authorities, agricultural authorities, environmental authorities, engineering consultancies or local councils.

C. What ongoing maintenance costs would this entail in the future?

This could be based on similar cases elsewhere, from sources such as those outlined in B.

D. Are the current and future costs of flood protection likely to exceed the damages that floods could incur if they are not checked? (see Damage Costs Avoided, above)

The type of land use that would be flooded as a result of the wetland function being lost may give a rough indication. The possible damage costs (see previous section) can approximate the benefits to be derived from maintaining the flood control function.

E. Will the flood protection measures themselves incur indirect costs or benefits?

There is frequently a 'risk-environment trade-off' (Fordham and Tunstall, 1990) between reduced risks of flooding through construction of flood defence measures, and impacts on local environmental quality. Such impacts might involve a loss of landscape amenity in terms of open spaces, river views and accessibility, or local ecosystem damage. There is also the potential that flooding could simply be increased elsewhere downstream. There could be **benefits** associated with protective structures, such as increased recreation possibilities or energy production potential, suggesting that construction costs cannot be attributed solely to achieving flood protection. This information is likely to derive from local experts and planners or previous cases elsewhere.

F2 Groundwater recharge

"The recharge of groundwater by infiltration and percolation of detained floodwater into a significant aquifer"

A wetland's ability to recharge groundwater will only be of value if that groundwater is then of some benefit to society. It is important to identify a subsequent use of groundwater, or perhaps non-use motivations for maintaining supplies of groundwater, for economic value to be associated with the wetland's recharge function. Again to assess the economic value of this recharge function, the following procedures must be used:

I) Assessing the potential for the recharge function of a wetland to influence groundwater.

II) Determining the extent to which the wetland influences levels of groundwater, and how these would be affected were the wetland to be removed (the "with"- and "without"- comparison).

III) Identifying the potential uses of groundwater – for example abstraction, maintenance of groundwater discharge functions and for future use.

IV) Estimating the economic value of the wetland's groundwater recharge function with respect to each of these "uses".

The benefits may be direct, such as abstraction of water for irrigation or domestic use, or indirect, such as the maintenance of water table levels. In addition to these use values, there may be non-use values of maintaining groundwater supplies. Non-use values can be attributed to the maintenance of groundwater supplies for subsequent generations, but only if use of the reserves is anticipated.

Valuation techniques

As far as the extractive uses are concerned, the techniques involved in assessing the economic value of groundwater recharge are much the same as those outlined for the 'groundwater discharge/surface water generation' function below. Studies that have considered values for groundwater supply are outlined as they illustrate techniques that may also be useful for assessing the value of surface water. They include hedonic pricing based on variations in availability of groundwater irrigation supplies; costs of establishing substitute well sites; and contingent valuation of willingness to pay for alternative piped water supplies. A number of studies have assessed values associated with maintaining the quality of groundwater (which may be relevant to the in-situ uses of the recharge function) and these are considered under the 'nutrient retention' function. Two other in-situ use values arising from groundwater recharge include prevention of land subsidence and salt water intrusion.

Prevention of land subsidence:

- *Hedonic property pricing.* Hedonic pricing is used to analyses a price differential in property that is attributable solely to the risk of subsidence. If identical sets of housing exhibit variation in prices, and the only non-constant attribute is the risk of subsidence, then price differences can be related to the buyers' willingness to pay to avoid subsidence. However, it is necessary to assess all relevant variables that could affect price (e.g. location, size, aspect, age of property etc), and to isolate the effect of subsidence from these.

- *Damage costs avoided*. Predominant land uses are identified and the various costs of a potential subsidence assessed. Estimation of the damage costs that are avoided due to ground water recharge provides an upper bound estimate of the value of this function as it does not technically value society's willingness to pay to avoid the subsidence. Instead, it values the full extent of costs expected to result from subsidence, which could exceed the cost of alternative measures that might be used to negate the economic impacts. However, it may not be feasible to estimate all the costs involved, particularly the intangible costs.

Salt water intrusion

- *Residual imputation and variants.* Intrusion of salt water can occur due to falling groundwater in levels in areas near to the coast. Salt water intrusion can render groundwater unusable for irrigation, thereby impinging directly on agricultural production. The change in net returns that this would cause can be used to assess the value of maintaining ground water levels to prevent the intrusion of salt water.

F.3 Groundwater discharge

'The upward seepage of groundwater to the wetland surface'

The discharge of groundwater contributes to the surface water within a wetland and to downstream flow. Whether water originates from direct precipitation, groundwater discharge or another source will not in itself influence the value attributable to a wetland's surface water generation. Groundwater discharge at a wetland is not considered as a separate function in terms of valuation. The discharge will contribute to surface water within the wetland and therefore can be valued with the same techniques as outlined in the section above. Characteristics of discharged groundwater, such as temperature and chemical constituents, might influence other wetland functions such as primary productivity and the ability to retain nutrient, but any benefits that may be associated with these will be included in the valuation of specific functions. Identifying the source of water could become important if the focus of valuation is extended to a level beyond the wetland – perhaps to the catchment level. In this case, benefits associated with surface water could be linked to its previous sources, thereby attributing value, perhaps, to ecosystems that have facilitated the previous recharge of groundwater elsewhere.

Surface water generation

'The discharge of groundwater into the surface water system'

Groundwater discharge and surface water generation can be considered as identical functions for the purposes of valuation. Whether water originates from direct precipitation, groundwater discharge or another source does not influence the value attributable to surface water generation. The surface water that is generated contributes to the stocks and flows of surface water, which support a variety of in-situ and extractive uses as well as non-use values.

Extractive uses of surface water include use of water for irrigation and domestic purposes. In-situ uses of surface water are more varied and can include maintenance of habitats and provision of aesthetic and recreational value e.g. chalk streams, which are BAP priority habitat. A number of the in-situ uses of surface water are also considered within other functions. For instance, the reliance of characteristic wetland ecology on surface water and anaerobic conditions resulting from inundation (and the subsequent capacity to retain excess nutrients) are considered under the 'ecological' and 'nutrient retention' functions, respectively. Downstream habitat and biodiversity maintenance are considered below only in so far as they might contribute to recreational and amenity value. Other benefits associated with maintaining biodiversity could be significant (for instance non-use values) and valuation methods for these are outlined under the subsection on 'ecological' functions.

Valuation techniques[2]

As mentioned earlier, the techniques outlined below for valuing extractive and in-situ uses of surface water are also applicable to the extractive uses listed under the groundwater recharge function. The main extractive and in-situ uses and possible techniques used to value them have already been comprehensively considered in, for example, Gibbons (1986); Young (1996); National Research Council (1997), Renzetti, (2002), and hence we only provide a few illustrative application examples here.

- *Market based transactions* Surface water abstraction for use in irrigation can be valued using market prices observed in rentals and sales of water rights. In order for traded water rights (either for use of water over a specified period or for a permanent right to water use) to reflect the economic value of water use, allocation and enforcement of property rights is required. If necessary, prices should be adjusted to reflect long term considerations (i.e. social values). In practice rental rates may be affected by factors other than the marginal value of water. Although observations of prices on markets for perpetual water rights are more appropriate for long run planning contexts, some degree of care is required in converting this capitalised asset value into the annual values conventionally used in planning and policy analysis (Young, 1996). Furthermore, the use of water right prices, in circumstances where crop prices are supported by agricultural subsidies, will lead to overestimation of the social value of irrigation water.

- *Residual imputation and variants.* This is one of the techniques that is most widely used to value irrigation water. It was employed by Ruttan (1965) in an early study

[2] It should be noted that the use of irrigation in the UK is supplemental to rainfall and that many crops are not irrigated (Weatherhead and Knox, 2000), however, where irrigation does take place the valuation techniques in this section are applicable.

that demonstrated the difference between the value of irrigated and non-irrigated agricultural production. Some degree of care has to be taken with its use to ensure that statistical problems, such as multicollinearity between variables, do not bias the analysis. Linear programming models have been applied to farm budget data to derive shadow values on irrigation water (e.g. Colby, 1989). Here, crop type is the most important determinant of the marginal value of irrigation water. The presence of uncertainty makes valuation of agricultural water use difficult, due to uncertainty in the need for irrigation (arising from climatic variation, for example) and in water supplies. Farmers' attitudes towards risk must therefore be considered when undertaking studies. Market distortions and externalities also need to be taken into account.

- *Derived Demand Functions.* This technique has been used to estimate households' valuation of domestic water supplies employing relatively easily acquired price and quantity data, for example in Young and Gray (1972) and Gibbons (1986). Though these studies address households' valuations of a given quantity of water, they did not address complications created by variations in water quality or service reliability. These issues have been considered in studies using contingent valuation and avoided cost approaches.

- *Hedonic pricing.* Hedonic pricing can, in principle, be used to derive the value of maintenance of river flows by surface water generation (Loomis, 1987). Individuals or businesses (including farms) might pay a premium for property located close to a river. It may be difficult to distinguish from use of water in the river from other locational factors, such as benefits associated with aesthetics, recreation or transportation that result from proximity to the river. It is also difficult to determine the contribution made by the discharge of ground water to maintenance of water levels in the river. Few studies have used hedonic pricing to value surface water generation, presumably due to these complications and demands of the technique. One of the few examples decomposes the value of agricultural land as a function of its attributes including the use of irrigation water (Faux and Perry, 1999).

- *Replacement cost/avoided cost.* Avoided cost has been used to value hydroelectric power generation (see Gibbons, 1986). The cost that would be incurred if the capacity to generate power was provided from an alternative source is used to impute the value of the hydroelectric power generated. However the method is problematic as it ignores the price elasticity of demand for electric energy. The approach can also be applied to valuation of surface water generation where water is abstracted to provide drinking water. The expense of finding an alternative water supply, though considerable is, likely to be exceeded by benefits of continued use of the existing source. The technique may be particularly suited to this application if there is difficulty in valuing the health implications of restrictions in water supply.

F.4 Sediment retention

'The net retention of sediments carried in suspension be waters inundating the wetland from river overbank flooding and runoff from the contributory area'

The benefits of a wetland's ability to retain sediments that settle out as the water velocity passing through the wetland decreases relate essentially to the reduced sediment load in waters downstream. However, the retention of sediments also contributes to other potential benefits **within** the wetland such as the support of ecological functions, as this deposition of sediments maintains inputs and improves water quality within the wetland. Sedimentation within a wetland maintains biodiversity and biomass harvesting and is thus closely linked to the ecological functions, the evaluation of this benefit will be examined within the context of the 'ecological functions' section.

Valuation of the damages resulting from sediment loading (or, benefits of reduced loading due to awetland's sediment retention function) is considered. Again, assessing the economic value of the sediment retention function of a wetland involves:

I. Assessing the potential for sediment to be retained by the wetland,

II. Determining the extent to which the wetland reduces the sediment load of the downstream flow and how this will be affected if the wetland were to be removed (*with* and *without* comparison).

III. Identifying any adverse effects of increased sediment loads that result in a loss in economic welfare; linking possible (quantified) increases in sediment load to these adverse effects

IV. Estimating the economic value of each of the benefits of reduced sediment loading due to the wetland's sediment retention function.

In order for the wetland to perform this sediment retention function, there must be sediment within the water entering the wetland and the conditions for the deposition of sediment must be met. The sediment load of water entering a wetland is dependent upon the upstream catchment, for example recent logging or intensive cultivation/soil compaction can lead to higher levels of sediment being released into the watercourse. For this sediment to be deposited, the conditions within the wetland must be such that the water is moving slowly.

Valuation techniques:

Retention of sediment reduces the load in water downstream and thereby improves water quality. The value of this may be most readily estimated in terms of the additional costs that would be incurred by industrial and municipal users of water through the necessity for water treatment in the absence of sediment retention. Higher water quality may also lead to increased opportunities downstream (e.g. for recreation and commercial fisherie) and will have biological impacts on survival of habitats and species. Habitats and biodiversity are considered here only in so far as they might contribute to recreational and amenity value. Techniques for valuing the benefits of improved water quality (or, conversely, the costs of poor water quality), are covered under the nutrient retention function.

Additional benefits of reduced sediment loads include mitigation of damages to water conveyance facilities e.g. through deposition of sediment in rivers, drainage ditches and irrigation canals, which can lead to adverse effects on navigation and water storage capacity and can increase flooding and costs of maintenance. Some of the techniques that may be used to value those benefits that have not already been considered under other functions are discussed below.

- *Avoided cost/damage costs.* The benefits of maintaining navigation can be estimated in terms of the avoided costs of alternative transport. This approach does not usually account for the differences in speed between alternative modes of transport. Alternatively, the benefits of maintaining navigation can be valued as the damage costs avoided in terms of reduced accidents and groundings. However, values are likely to be low, especially if the costs of infrastructure have already been accounted for. The benefits of mitigating damages to water conveyance, such as deposition in drainage ditches and irrigation canals, is peculiar to the sediment retention function. Estimating the damage costs avoided in terms of the costs of reversing possible adverse impacts, is the most appropriate valuation technique to use.
- *Residual imputation and variants.* The presence of fine silt particles in water used for irrigation can lead to a loss in productivity, as they can seal the surface of the soil, making it impermeable. However, the addition of sediment can also increase soil fertility and thereby improve productivity. As sediment impinges directly on agricultural production, for which market prices exist, then changes in marketed outputs can be used to assess the value of sediment retention.

F. Valuation of Biogeochemical functions

F.1 Nutrient retention

"The storage of excess nutrients (nitrogen and/or phosphorus) via biological, biochemical and geochemical processes in biomass (living and dead) and soil mineral compounds of a wetland"

In storing nitrogen and phosphorus, the nutrient retention function has the effect of improving water quality downstream of the wetland. However, the retention of nutrients contributes to other potential benefits **within** a wetland such as the support of ecological functions, as these nutrients contribute to high levels of productivity. As this productivity contributes to biodiversity maintenance and biomass harvesting, it is closely linked to the ecological functions and will be evaluated in that section.

The assessment of the economic consequences of damage to wetlands relating to this nutrient retention function involves the following steps:

I) Assessment of the potential for nutrients species nitrogen and phosphorus to be retained by the wetland.
II) Determination of the degree to which nutrient release into water sources is reduced by the wetland (with and without comparison)
III) Identification of potential adverse effects of increased nutrient levels and where they might occur
IV) Assessment of the degree to which an increase in nutrient levels that results in adverse effects represents a loss in economic welfare

As the nutrient retention function refers to the *storage* of nutrients within a wetland site a further stage of assessment is required. Since retention implies a comparatively short time horizon, if no subsequent **removal** of these nutrients from the wetland occurs, then potential thresholds of nutrient levels and the limited time frame before higher nutrient levels once again enter the water-stream, must be considered. The permanent removal of nutrients from the wetland is discussed in the next section on "nutrient export". Presumably, if no export occurs, then continual nutrient retention cannot be sustained indefinitely. This could be due to impacts on the wetland itself or an overloading of the capacity of the wetland to perform the function, thereby crossing some threshold beyond which the function is either degraded or is not able to process any further nutrients. Where the function is sufficiently degraded, it may be possible that increased levels of nutrients are released as a result of additional recourse to this function. Where nutrient retention dominates, without any subsequent export, it might be appropriate to consider concepts such as critical loads, sustainability, and maintaining safe minimum standards in addition to economic valuation of the function.

Thus, a further stage of analysis involves questions as to the physical capacity of a wetland to continue to absorb nutrients:

V. Determination of whether a possible threshold level of nutrients above which the wetland function is over-burdened exists, and whether current or predicted future nutrient levels threaten to cross such a threshold.

Assessing the potential for retention

The function of nutrient retention is only performed when excess nutrients entering the wetland ecosystem, where excess nutrients refer to a level of nutrients that would not be expected under natural conditions. To determine whether nutrients are being retained within a wetland depends on certain conditions being met. Input of nitrogen, ammonia or phosphorus are an essential pre-requisite whether through a direct or indirect source, with other factors including vegetation type, soil-water regime and the pH of soil, depending on which process is influencing the retention of nutrients.

Determining the reduction in nutrient release

It is essential that the quantities of each of the nitrogen, ammonia and phosphorus nutrient inputs to the wetlands are identified. Once in a wetland, nutrients can be retained through storage either in living (plants) and dead (soil organic matter) biomass, through biogeochemical interactions with the mineral component of soil; and through the deposition of particulate matter within the wetland (Figure 3). It is possible for a wetland to store nutrients in all these ways or any combination. Thus it is essential to determine the nutrient retention capacities and how much it is being utilised for each possible means of storage within the wetland.

Identifying adverse effects of increased nutrient levels

The impact on aquatic ecosystems of nutrient enrichment is no different to the impact on the wetland ecosystem, in that biological productivity will increase. However, although wetlands are typically able to withstand substantial increases in the concentration of available nutrients, many other aquatic habitats are not nearly so tolerant. While the increased biological productivity within aquatic ecosystems can be beneficial, if the capacity of an ecosystem to assimilate the nutrient-enhanced productivity is exceeded, water quality degradation will occur with detrimental impacts on the components of an ecosystem and ecosystem functioning, and consequently human welfare. Caddy (1993) demonstrated the effect of this assimilation threshold for fisheries, where a positive relationship exists between yield and nutrient loading until a maximal point, after which the fisheries yield declines as nutrient load increases.

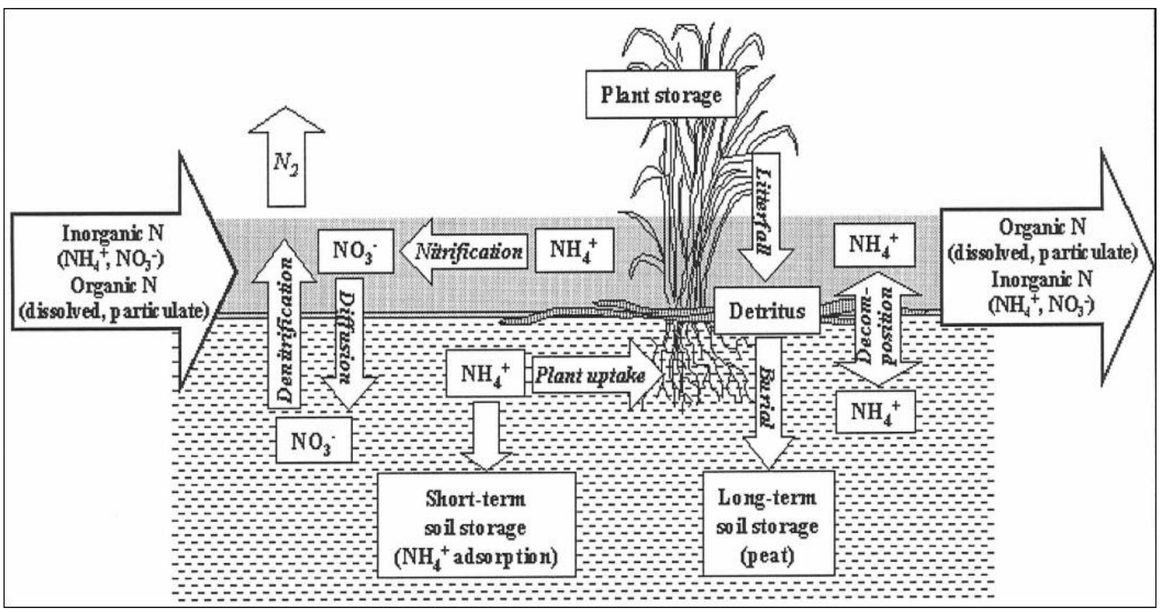


Figure 3: Summary diagram of the fate of nitrogen entering a wetland. (From DeBusk, 1999)

Effect on human welfare

Improvements to water quality will only be of value if human welfare is affected. The effects of eutrophication and acidification of surface waters have the following impacts on human welfare:

- Increased vegetation impedes water flow and movement of boats
- Loss of habitats and biodiversity
- Water is unsuitable for drinking, even after treatment
- Decrease in the amenity value of water, e.g. for water sports
- Disappearance of commercially important species

Thus, the nutrient retention function performed by a wetland produces economic benefit in terms of navigation, biodiversity and water quality downstream of the wetland. The navigation of waterways downstream of the wetlands is covered in the section on "sediment retention", and downstream habitat and biodiversity maintenance are considered here only so far as they might contribute to recreational and amenity value.

The benefits of wetlands in reducing water pollution can be classified and therefore valued as (Freeman, 1982):

Recreation - such as fishing, boating, hiking and aesthetic appreciation of the water-body or waterside site;

Non-use benefits - from knowing that water quality and ecosystem health are maintained for the sake of others;

Diversionary uses - including health aspects associated with abstracting drinking water, costs associated with treatment of municipal water supplies, costs to households of possible corrosion of pipes and appliances, and treatment costs of water used in industrial processes and cooling;

Commercial fisheries - whose productivity may be heavily affected by levels of pollutants.

Valuation techniques:

The main impact of storage of nitrogen and phosphorus is improved water quality; thus this function is discussed here with respect to water quality. A few illustrative examples of valuing benefits of improved water quality are outlined below. Impacts on recreation can be valued using the travel cost method. The benefits for drinking water supplies can be considered, via defensive expenditures. Nutrient retention benefits can be considered generally in terms of the costs of providing substitute treatment facilities. Potential increases in the costs of industrial production processes are not considered here. The residual imputation methodology, discussed with reference to the 'surface water generation' function, could also be employed to value the benefits of improved water quality as an input to production processes.

- *Contingent valuation.* CV based research has been widely used to consider water quality. Jordan and Elnagheeb (1993) used the approach to assess households' valuations of improvements in drinking water supplies (due to reductions in nitrate levels). One of the most challenging aspects of using this approach is the manner in which water quality information is conveyed to survey respondents, and specifically whether objective or subjective measures of water quality are used. Poe (1998) argues that objective measures are preferable because people do not have reliable and well informed reference points. Conjoint analysis and contingent ranking has also been used to value water quality improvements. For example, Georgiou et al (2000) used contingent ranking to value urban river water quality improvements. The annex contains a case study that examines water quality in the Philippines

- *Travel cost.* The travel cost method has been used to assess the value of improved water quality at recreational sites. A complex form of travel cost analysis, which includes measures of water quality as independent variables, is applied to sites which vary in water quality (but are similar in other attributes) or to one site for which water quality changes over time. This is an extremely involved procedure, which measures only the recreational benefits associated with improved water quality downstream. There are a number of difficulties with such analysis. In particular, for a multi-site study, the influence of water quality between sites needs to be isolated from other varying attributes that might affect recreation demand. In the case of a single site temporal study, changes in water quality need to be isolated from other attributes that might change over time. Smith and Kaoru (1990) undertook a meta analysis of travel cost studies that relate to water-based recreational values. They found that the following five features consistently had an influence on results: type of recreational site, the definition of a site's usage and quality, measurement of the opportunity cost of time, the description of substitutes, and specification of the demand model.

- *Hedonic price method.* The price of properties in close proximity to water bodies can be affected by the quality of the water and therefore by nutrient retention. The value of the nutrient retention function is derived from (a) property values that are attributable to water quality and (b) the role of the function in maintaining the water quality. This entails analysis of prices for otherwise similar properties that are located close to polluted and unpolluted water bodies. The data demands are, however, considerable. A rough approximation of value can be derived directly from a summation of adjustments in property prices, which could be based on assessments of experts, such as estate agents, rather than actual observed price differentials in the property market.

- *Defensive expenditures/avoided cost.* The value of improved water quality can be estimated based on the expenditures undertaken by people to avoid consumption of poor quality water. The sum of defensive expenditures on marketed goods such as water purification equipment represents the lower boundary on society's willingness to pay for improved water quality. This accounts only for changes in behaviour made by consumers in response to poor water quality. It does not take into account consumers who do not undertake defensive actions but would nonetheless prefer improved water quality. Such individuals may be inhibited from acting by inconvenience associated with the defensive activities, or lack of information about pollutant levels and possible adverse effects. Abdalla et al (1992) use this valuation approach to determine the time and money that households expend to avoid risk arising from groundwater contamination. Their approach assumes that households undertake a two-step decision-making process in which they first decide whether to undertake any avertive action, and then decide on the intensity of those actions. Adalla (1994) also provide a survey of the literature on averting cost methods.

- *Replacement cost.* The retention of nutrients can be valued using the replacement cost in terms of the cost of substitutes. Substitute activities include reduction of nitrate and phosphate pollution at source by limiting applications of agricultural fertilisers or the installation of water treatment facilities. The replacement cost is particularly useful for situations where the benefits of reduced nutrient loading are difficult to estimate. Examples include estimation of the benefits of avoiding deleterious health effects or the benefits of maintaining water quality and ecosystems for future generations.

F.2 Nutrient export

"The removal of excess nutrients (nitrogen and/or phosphorus) from a wetland via biological, biochemical, physical and land management processes"

The sources of value attributable to nutrient export are the same as for nutrient retention (previous section). The important difference, however, is that **export** implies a permanent removal of the nutrients, while **retention** suggests that nutrients might once more enter the water stream. The level of nutrients that can be diverted from the water stream by a wetland may alter according to which of these two functions is being performed.

The initial valuation techniques for improvements to the water quality will be the same for both nutrient retention and export (hence, the section 'Nutrient retention' should be referred to for valuing the nutrient export function). The same initial stages of analysis should be undertaken to assess the economic consequences of damage to the wetlands relating to this nutrient export function:

I Assessing the potential for the nutrients, nitrogen and phosphorus, to be retained by the wetland.

II Determination of the degree to which nutrient release into water sources is reduced by the wetland (with and without comparison).

III Identification of potential adverse effects of increased nutrient levels and where they might occur.

IV Assessment of the degree to which an increase in nutrient level that results in these adverse effects represents a loss in economic welfare.

Assessing the potential for nutrient release to be reduced

Within the nutrient export function, there are three major processes by which the nutrients are removed or "exported" from wetlands: via gaseous export of nitrogen, export through land management and physical processes (Figure 3).

Nitrogen can be released into the atmosphere through bacterial reduction to either nitrous oxide or atmospheric nitrogen (Mitsch and Gosselink, 2000). If nitrogen is present in the form of ammonium or ammonia, this can be released to the atmosphere through volatilisation. The uptake of nutrients by plants increases ecosystem productivity, and the nutrients can be removed through land use management practices. Land-use practices may export the nutrient directly, through the burning of wetland vegetation (loss to the atmosphere), harvesting of vegetation (e.g. crops for consumption), and harvesting of the produce of animals (which have assimilated the nutrients through the consumption of vegetation). Further uses of the land that could lead to the export of nutrients are forestry and hunting, fishing and shooting. The final method of export is when nutrients are removed from the wetland by water or wind transport processes. While the nutrients are permanently removed from the wetland in question, they can then act to support other ecosystems or as a source of nutrient contaminants for other ecosystems.

Determining how actual nutrient release is affected by wetlands

The quantities of nitrate, ammonia and phosphorus entering the wetland must be identified. In addition to determining how much of each nutrient is retained, further analysis is required to determine whether nutrients previously stored within a wetland are now being exported, and if so in what quantities as this could adversely affect water quality and this function could represent a *cost* rather than a benefit. This is a direct consequence of the export function releasing nutrients back into the aquatic ecosystem. Otherwise the means of analysis are the same as for the nutrient retention function.

Identifying adverse effects of increased nutrient levels

In reducing the levels of nutrients entering waters downstream of the wetland, this function has the same effects as the previous function. However, a further consideration exists over where the nutrients ultimately end up, having been exported from the wetland. Will the means of export simply transfer the problem elsewhere? The value attributable to the wetland for such export should take account of any such 'external' effects that might result off-site. For example, a consequence of denitrification could be increased environmental pollution, with nitrous oxide released into the atmosphere being oxidised to nitric oxide, which is a contributor to acid rain. Also, as the export through land-use management involves biomass harvesting, these values are closely linked to the ecological functions and will be evaluated in that section.

Valuation techniques:

Again, the same methods as used in the previous function are used for the effects on water quality, except that a release of nutrient will decrease water quality downstream of the wetland.

F.3 Trace element storage

"The storage of trace elements via biochemical and physical processes in the mineral compounds of the wetland soils"

Trace elements (including heavy metals) are natural constituents of the Earth's crust and are present in varying concentrations in all ecosystems. At trace levels, many of these elements are necessary to support life. The essential metals include cobalt (Co), copper (Cu), iron (Fe), manganese (Mn), molybdenum (Mo), vanadium (V), strontium (Sr), and zinc (Zn). However, at elevated levels they become toxic, and become significantly detrimental to living organisms. They are stable and persistent environmental contaminants since they cannot be degraded or destroyed. They tend to accumulate in soils, seawater, freshwater, and sediments. Excessive levels of metals in the marine environment can affect marine biota and pose risks to human consumers of seafood. Non-essential heavy metals of particular concern to surface water systems are cadmium (Cd), chromium (Cr), mercury (Hg), lead (Pb), arsenic (As), and antimony (Sb).

Sources of trace elements found in watercourses include fertiliser (organic and inorganic) input, aerial input through atmospheric or gaseous industrial discharge and through organic industrial waste or sewage sludge. Trace elements can enter the wetland either through groundwater discharge, run-off from slopes, atmospheric deposition and from river water entering the wetland through over-bank flooding.

This storage function has the effect of improving water quality downstream of the wetland, and is thus treated in a similar to fashion the nutrient retention function. The assessment of the valuation of trace element retention follows the same five processes as for nutrient retention, since there is the potential for trace elements to be released if thresholds are exceeded. For further detail refer to the nutrient retention function:

- I Assessing the potential for trace elements to be retained by wetlands.
- II Determination of the degree to which trace element concentrations in water sources are reduced by wetlands (with and without comparison).
- III Identification of potential adverse effects of increased trace element concentrations and where they might occur.
- IV Assessment of the degree to which an increase in trace element concentrations that represents a loss in economic welfare.
- V Determination of whether a possible threshold level of nutrients above which the wetland function is over-burdened exists, and whether current or predicted future nutrient levels threaten to cross such a threshold.

Assessing the potential for trace element concentrations to be reduced

Within a wetland, trace elements can be retained through either physical or biogeochemical means. If the conditions are right for deposition of sediments, for example, when the wetland is inundated with polluted water, trace elements can be deposited in the solid form of particulate matter. Trace elements are bound biogeochemically within the soil, and are thus immobilised. Whether this process occurs depends on the speciation of the elements themselves and the soil conditions.

Determining the reduction in trace element concentrations

It is essential that the trace elements entering the wetlands are identified and quantified. Once in a wetland, the trace elements can be stored through two major processes of physical and biogeochemical retention. It is possible for a wetland to store trace elements in either or both of these methods. Thus it is essential to determine the capacity of the wetland to store trace elements by each possible means of storage within the wetland.

Identifying adverse effects of increased trace element concentrations

In reducing the concentrations of trace elements entering the waters downstream of the wetland, this function has the effect of improving water quality. The impact of increased levels of trace elements within the aquatic ecosystem is in bioaccumulation, where the trace elements enter the food chain through uptake in living plants. Concentrations of the elements increases the further up the food chain, with an increase in the toxicity resulting in a decrease in the survival, size and density of some fish, and the loss of other fish and aquatic biota from lakes and streams.

As the ability of the trace elements to be deposited with other sediments suspended in the water flow is not constrained to wetland areas, in other areas of the watercourse where sedimentation occurs, trace elements will be deposited and their effect may be limited. However, in areas where flow rates are high or the water is constantly disturbed, for example in streams and rivers in the upper reaches of a catchment and in tidal estuaries, the concentration in the water column will be high as the elements do not settle. Additionally, sediments previously deposited may be disturbed, releasing further trace elements causing high levels of toxicity.

Valuation techniques:

The ecological changes resulting from contamination of water by trace elements impact human populations by changing the availability of seafood and creating a risk of consuming contaminated fish or shellfish; reducing our ability to use and enjoy our coastal ecosystems; and causing economic impact on people who rely on healthy coastal ecosystems, such as fishermen and those who cater to tourists. Thus, the same methods of evaluation used for valuing water quality within the nutrient function can be applied here.

F.4 Trace element export

"The removal of trace elements from a wetland via biological, biochemical and physical processes"

The sources of value attributable to trace element export are the same as for trace element storage. The important difference however is that **export** implies a permanent removal of the trace elements, while **retention** suggests that trace elements might once more enter the water stream. The level of trace elements that can be diverted from the water stream by a wetland may alter according to which of these two functions is being performed.

The initial valuation techniques for improvements to the water quality will therefore be the same for both trace element storage and export. The 'Trace element storage' section should be referred to for valuing the trace element export function. The same initial stages of analysis should be undertaken to assess the economic consequences of damage to the wetlands relating to this trace element export function:

I. Assessing the potential for trace elements to be retained by a wetland.
II. Determination of the degree to which trace element concentrations in water sources are reduced by the wetland (with and without comparison)
III. Identification of potential adverse effects of increased trace element concentrations and where they might occur
IV. Assessment of the degree to which an increase in trace element concentrations that results in these adverse effects represents a loss in economic welfare

Assessing the potential reduction in trace element release

Trace element export represents a closely linked function affecting water quality in interaction with trace element storage. There are three major processes by which trace elements are removed or "exported" from a wetland: via the uptake of trace elements by plants, and the physical and biogeochemical remobilisation of the elements. The remobilisation of trace elements can either happen in a physical manner, where the trace elements retained in the particulate matter re-enter the drainage system when the soil in which they are deposited is eroded. Alternatively, biogeochemical remobilisation occurs when the chemical bonds between the trace element and the soil break and the trace elements then re-enter the watercourse.

Determining the actual effect on trace element concentrations

It is essential that the trace elements entering wetlands are identified and quantified. A direct consequence of the export function is that trace elements can be released back into the aquatic ecosystem or can enter the food chain. Analysis is required to determine whether trace elements previously stored within the wetland could, with a change in the wetland, shift to being exported, and if so in what quantities, as this could adversely affect water quality. In this way, this function could represent a *cost* rather than a benefit.
An assessment is required into the trace elements exported from the wetland through plant uptake as this not only has implications for the food-web; it is the only means by which the elements are actually removed from the wetland.

Identifying adverse effects of increased trace element concentrations

In reducing the levels of trace elements entering the waters downstream of the wetland, this function has the effect of improving water quality. The impact of increased levels of trace elements within the aquatic ecosystem is in bioaccumulation, where the trace elements enter the food chain through biological uptake into living plant matter. These bioaccumulation effects are not constrained to the aquatic ecosystem due to the export of the trace elements through plant uptake, thus the ecological functions of biomass maintenance and harvesting can be affected by increased concentrations of heavy metals.

As the ability of the trace elements to be deposited with other sediments suspended in the water flow is not constrained to wetland areas, in other areas of the watercourse where sedimentation occurs, trace elements will be deposited and their effect will be limited. However, in areas where flow rates are high or the water is constantly disturbed for example streams and rivers in the upper reaches of a catchment and tidal estuaries, the concentration will be high as the elements do not settle. Additionally, sediments previously deposited may be disturbed, releasing further trace elements, and causing high levels of toxicity.

Valuation techniques:

The ecological changes resulting from contamination of water by trace elements impact human populations by changing the availability of seafood and creating a risk of consuming contaminated fish or shellfish, reducing our ability to use and enjoy our coastal ecosystems, and causing economic impact on people who rely on healthy coastal ecosystems, such as fishermen and those who cater to tourists. Thus, the same methods of evaluation used for valuing water quality within the nutrient function can be applied here.

F.5 In situ carbon retention

"The retention of carbon in the form of partially decomposed organic matter or peat in the soil profile due to environmental conditions that reduce rates of decomposition"

Where hydrological and geomorphological conditions combined with appropriate climatic conditions create wet soils and a low rate of decomposition, partially decomposed organic matter may accumulate within the soil in the form of peat deposits. This leads to the storage of the carbon bound in organic matter, which is a major benefit of a wetland's ability to accumulate peat. Ecosystems that can act as carbon 'sinks' are becoming increasingly important, as the link between carbon compounds in the atmosphere and global warming becomes apparent (Maltby et al., 1993). The accumulation of peat can create a substrate for rare and unique plants and animals, which is closely linked to the ecological function of biodiversity maintenance and will be evaluated in this context in that section.

The assessment of the economic value of *in situ* retention of carbon through the accumulation of peat requires the degree of accumulation and storage to be estimated, along with their significance in terms of global carbon emissions. The following assessment procedures are required:

Assessing the potential for the wetland for peat accumulation and carbon sequestration

I Determining the extent to which a wetland acts as a carbon store (the “with”- and “without”- comparison).

II Identifying the potential uses of accumulated peat

III Estimating the economic value of the *in situ* carbon retention function

Assessing potential for peat accumulation

For wetlands the accumulation of carbon may be offset by the simultaneous production of the greenhouse gas methane. In order to ascertain whether a wetland is a carbon sink/source/neutral with respect to greenhouse gas emissions, there is a need to know whether the wetland emits more methane than it stores as organic carbon. Although data on the amount of organic carbon stored by a wetland is easily generated, it is difficult to calculate the production of methane, and as such, little data on wetland methane production currently exists for the UK. While it is known that emissions of methane are lower from high salinity and high altitude wetlands, Parkes (2003) demonstrates how the use of different estimations of methane emissions can lead to substantially different assertions as to whether a wetland is a greenhouse gas sink or source. Parkes found that for the Wash saltmarshes, Eastern England, the use of estimates of methane fluxes at the lower end of those reported lead to the conclusion that saltmarshes act as a carbon store, while the use of high methane fluxes found the saltmarshes were acting as a source of carbon. Due to these difficulties, and the range of conditions that can influence methane production, it is impossible to generalise which wetlands in the UK will act as carbon stores and which will act as sources. However, when peat wetlands do act as stores of previously accumulated peat this may be associated with a considerable value in terms of containing a potential source of atmospheric carbon. Conversely, if a wetland acts as a source of methane then this will be associated with a considerable cost given its effects as a greenhouse gas.

Extent of peat accumulation

The conversion of a peatland to another use would result in a considerable release of carbon. Peat is an excellent growing medium; however it requires drainage and cultivation to be used as pasture or for crops to be grown. The drainage of this peatland leads to increased run-off cutting into the peat layers, with the resulting drying-out as the peat is exposed enabling the carbon to be oxidised to carbon dioxide, which is ultimately released to the atmosphere. This is in addition to the destruction of habitat. If excessive drainage occurs at the edges of a peat bog, this will affect the continued accumulation of peat and lead to a change in the species formulation.

Uses of accumulated peat

While the slow rates of current accumulation might not generate significant value, the continued storage of carbon could be associated with considerable economic benefits.

Alternative uses of the accumulated peat exist, providing a source of fuel or horticultural materials. However, these are extractive uses of peat bog and due to the considerable length of time that it takes peat to accumulate, any benefit being attributed to the accumulation function based on extraction, is ruled out within the temporal framework of an economic study. While these uses are in conflict with the use of peat bogs as a store of carbon and represent a wholly unsustainable degradation of the ecosystem, they do provide a source of value that must be taken into account in any valuation of the wetland ecosystem – if only to ensure that negative economic and social values are considered.

Valuation techniques

The value of peat accumulation as a store of carbon is not easily assessed. Estimating the benefits associated with accumulating and maintaining a carbon store will inevitably represent an extremely rough approximation, the main reason being that the benefits of global warming mitigation are truly global, making their accurate estimation a complex exercise. Damage cost estimates vary between £7 and £70 per tonne of carbon (depending on assumptions about adaptive behaviour); the carbon permit market also provides a possible source of economic data in that a market price for carbon reduction permits is becoming established, prices are ranged from £2.50 to estimates around £10-13 tc.

F3. Valuation of ecological functions:

Ecological functions relate primarily to maintenance of habitats within which organisms survive. In terms of economic benefits, habitats and their overall structure are not themselves of value. It is the diversity of species (and their genes), landscapes, and services that are assigned economic value. It is therefore the biodiversity that is supported by a wetland that forms the basis for valuation of ecological functions. An example of the relationship between individual wetland ecological functions and economic valuation is illustrated in Figure C3, and is outlined below. Essentially, there is considerable overlap in the list of 'Ecological Functions'- so, for instance, biomass exports (whether via abiotic or biotic media) are necessarily dependent upon biomass production (allowing for any biomass import). Therefore, to avoid double counting, economic valuation will concentrate on only a limited number of ecological functions. It is biodiversity within, or supported by, a wetland that is most clearly linked to economic value, whether in maintaining it or in harvesting it. In terms of deriving the total economic value of the ecological functions of a wetland, estimation focuses specifically upon biodiversity maintenance and anthropogenic harvesting of biomass.

The schematic of Figure C3 shows the links between the various ecological functions that may be performed by a wetland. These are seen to derive ultimately from biomass production and food web support, although economic value is related to the 'end product' of biodiversity within (and beyond) the wetland. Biomass production, deriving ultimately from utilisation of solar energy, is a fundamental process within any ecosystem. However, a simple measure of the quantity of biomass (or perhaps the amount of embedded energy), does not give an indication of whether or not given **qualities** of biomass - the way in which it is structured and organised - or its physical location, may be of use or concern to humans. Habitat structure, that serves to encapsulate biomass production and all of the goods and services that may derive from it, is also not of direct use or concern to humans. The system framework, within which an ecosystem functions, may be attributed some form of primary or prior value, although it is not clear that this can be meaningfully quantified. Biodiversity, on the other hand, can be directly linked to satisfying human wants and needs. On-site biodiversity may be associated with non-consumptive use values (such as recreation) or non-use values (such as a bequest for future generations). Direct (anthropogenic) harvesting of the biota can provide consumptive use benefits (such as in providing building materials or foodstuffs). Wetlands that act as habitat microsites might also be valued for their biodiversity, although the degree to which the value of each species can be attributed to the wetland will depend on their reliance upon the wetland. Abiotic export and non-anthropogenic harvesting are not in themselves functions that are of value. However, depending on the extent to which this contributes to benefits to be derived from ecosystems elsewhere, some economic value might be attributable to the wetland as a result of such export.

Economic valuation of ecological functions performed by wetlands will therefore concentrate on the specific functions of biodiversity maintenance and anthropogenic biomass harvesting, although each of the other functions under the ‘ecological’ heading will be considered briefly in turn.

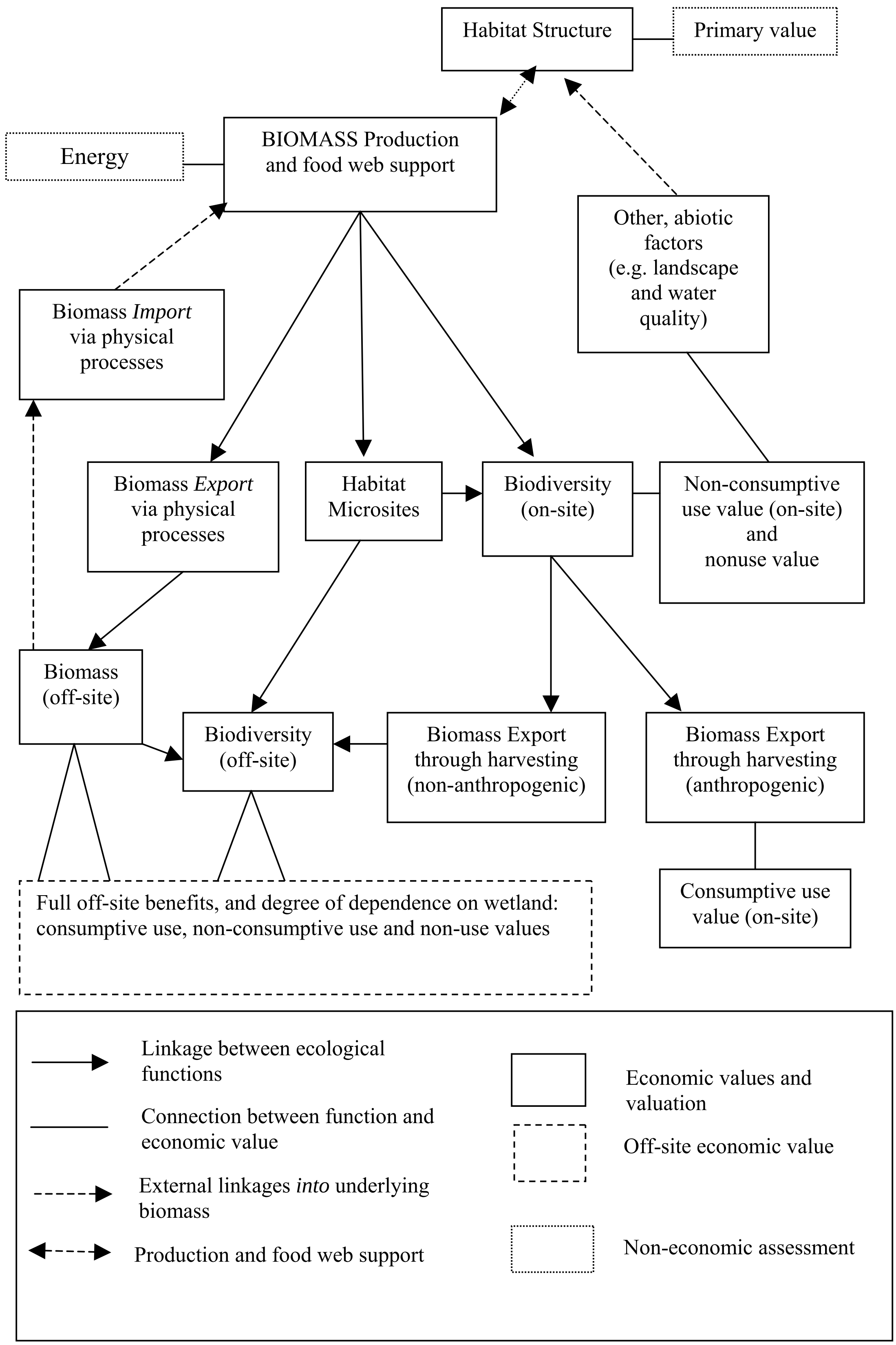


Figure F3. Ecological Functions and Economic Value

F3.1 Ecosystem maintenance

"The provision of habitat for animals and plants through the interaction of physical, chemical and biological wetland processes"

The function of ecosystem maintenance which is composed of three processes:

- Provision of overall habitat structural diversity
- Provision of microsites
- Provision of plant and habitat diversity

However, it is only through contact with, or concern for, the biological organisms which make up an ecosystem, that economic value is generally derived from the ecological functions that a wetland performs. It is thus 'biodiversity' within the wetland, the (anthropogenic) export of this biodiversity (*biomass export)* and the wetland's contribution to biodiversity elsewhere, that form the basis of the valuation of a wetland's ecological functions. While a wetland's biodiversity may derive ultimately from the process of *biomass production* and food-web support and is dependent upon overall ecosystem health and habitat structure (*provision of overall habitat structural diversity),* these processes are not in themselves of economic value to society. It is therefore on aspects of a wetland's biodiversity (both quantity and variety of organisms) that economic valuation is focused. A brief overview of the first two processes will be given below:

Provision of Overall Habitat Structural Diversity

The overall structure of the ecosystem and its habitats is clearly fundamental to continuing ecological processes and interactions. However, as with biomass production, habitat structure is not in itself a source of economic value. A possible caveat to this is in its contribution to the overall landscape, from which amenity benefits may be derived, although this will be included with the benefits to be derived from a wetland's biological features (*Biodiversity maintenance*). It is possible that value might be associated with habitat or ecosystem structure: the so-called primary value (see section 2.2.1.2). Such a consideration might suggest employing some form of safe minimum standard (Bishop, 1978), which recommends favouring preservation when confronted with irreversible damage, unless the benefits to be derived from the damaging action are considered to be too great to forgo. The concept of incorporating other such criteria into an economic evaluation framework is considered in previous sections.

Provision of Microsites

A wetland's provision of habitat microsites for feeding and breeding can generally be valued in the same manner as 'biodiversity maintenance'. However, in the case of habitat microsites, it is explicitly acknowledged that there is only a partial dependence of species on the wetland, so, for instance, any recreational and amenity value they might provide may be restricted to a particular (perhaps seasonal) time frame.

A number of studies have sought to value the benefits of maintaining habitats specifically as breeding microsites. For instance, Boddington (1993) claims to estimate the existence value associated with providing nesting habitat for wading birds, based on the difference in gross margins that are derived from farming (that is conducive to this nesting) compared with what the maximum returns might be under alternative management regimes. Cummings *et al.* (1994) employ continent valuation to estimate willingness to pay to preserve habitat that acts as a breeding site for the endangered Colorado Squawfish. Hanley and Craig (1991) refer to the importance of the Scottish Flow wetlands as a breeding site for birds in their contingent valuation study of use and non-use benefits for the area. Loomis (1987b) estimates willingness to pay to preserve a wetland habitat using a contingent valuation survey, one of the main advantages highlighted was maintaining a site for gull nesting.

While these studies focus specifically on the value of habitat as a breeding site, the valuation techniques and means of apportioning total value between alternative possible microsites - a number of which may be important for the survival of species - are identical to those outlined in '*Maintaining Biodiversity*'. It is this section that should therefore be referred to for estimating values associated with habitat microsite functions, paying particular attention to the sub-sections on 'Assessing a wetland's contribution to biodiversity maintenance' and 'Production Function Analysis'.

In assessing the economic value of biodiversity maintenance the following steps are required:

I Assessing potential sources of value from maintaining biodiversity

II Determining a wetland's contribution to biodiversity maintenance

III Identification of impact on human welfare

IV Estimating the economic value of the wetland's flood control function

Assessment of sources of value

Activities that result in the deliberate export or removal of material are dealt with in the next section on 'Biomass export through harvesting', and relate to 'consumptive use values'. Values associated with maintaining rather than harvesting biodiversity will be associated with non-consumptive use (based on 'aesthetic' benefits such as enjoying scenery or bird watching, and 'distant use' benefits such as reading magazine articles or watching television programmes that involve wetland biodiversity) or non-use values (for instance in preserving natural heritage for future generations). Any indirect use values, which derive from the services provided by a wetland, are included in sections relating to hydrological and biogeochemical functions (sections 1. and 2.).

Determining a wetland's contribution

It may be that some types of biodiversity within the wetland that are of value do not depend entirely upon the wetland. So, for instance, in the case of endangered species, there may be substitute sites available with suitable habitat, or species may be only partly reliant on the type of habitat that the wetland provides. This dependence will influence the extent to which the value of biodiversity maintenance can be attributed to the wetland. One possible management (and valuation) approach might involve relocating organisms to alternative sites. Another might be providing replacement or substitute sites where there is suitable habitat. The degree to which the wetland is integral to the preservation of biological attributes will clearly be a matter of scientific enquiry.

Dixon (1989) demonstrates how a wetland can provide a habitat for species that are then utilised off-site; although this survey of 'indirect products' was undertaken for mangroves, a wetland habitat type not found in the UK, it nevertheless highlights the range of indirect products that a wetland can provide. Dixon found that mangroves can provide important habitats for a range of fauna, including fish, crustaceans, molluscs, bees, birds, mammals and reptiles, while the benefits derived from these species, either for consumptive use or for recreation, are often derived outside of the ecosystem. Part of the value derived from these species is therefore attributable to the wetlands that provide an occasional, but often essential, habitat.

Identification of impact on human welfare

Possible indicators that a wetland's biodiversity may be of value to society

Non-consumptive use values:

- Recreation activities such as birdwatching, hiking, boating
- Official designation as a site for recreational purposes, perhaps as a national park

Non-use values:

- Recreational and amenity interest in scenic beauty or specific wildlife could indicate value attributed both by users and non-users to the continued existence of these features based on non-use motivations such as for the sake of future generations.
- Official designation as a protected area based on unique natural features may well indicate social value not dependent on any 'use' being made of the area.
- If the area is known to be home to unique or endangered species, society may well attach value to ensuring the continued existence of these species for their own sake.

Ongoing or historic public (or possibly private) projects designed to enhance or protect biological features might also be an indication of either amenity or non-use benefits deriving from such features.

Valuation techniques:

The economic values deriving from biodiversity maintenance are potentially considerable and could be an important component of a wetland's 'total economic value'. However, more so than other (hydrological and biogeochemical) wetland functions, maintaining biodiversity and healthy ecological functioning have the potential to be associated with highly uncertain, irreversible and ethically charged outcomes. The loss of species, genetic information or even whole habitat structures may be truly irreversible and the consequences in terms of ecosystem stability and future economic benefits may be wholly uncertain, with the potential for large impacts on the well-being of future generations. In addition to any ethical concerns for other species, moral and ethical issues relating to intergenerational equity are therefore raised when impacts upon biodiversity are threatened. This would suggest that other criteria than simply economic efficiency should be considered, such as applying sustainability constraints or safe minimum standards

Given the extreme uncertainty that surrounds both the complex interactions within and between ecosystems, as well as the benefits that humans could derive from them - now and in the future - there is a strong argument for supplementing purely economic analysis to allow for such considerable uncertainties. Therefore, particularly with regard to maintaining biodiversity and the linkages within ecosystems, issues such as ensuring sustainability of a given function, preserving critical components of 'natural capital', and maintaining 'safe minimum standards' of species populations and habitat requirement are extremely important.

Where the benefits of maintaining biodiversity are difficult to quantify and/or we can assume that the benefit of maintaining biodiversity is greater than the cost, or goals or targets have been set by alternative methods, then cost effectiveness analysis is a useful way of appraising management options.

With respect to biodiversity maintenance the valuation techniques of relevance include:

- *Contingent valuation.* Contingent valuation can be particularly useful for assessing the value of biodiversity maintenance, indicating willingness to pay for conservation of biodiversity. Contingent valuation is the only technique currently regarded as suitable for estimating non-use values associated with the maintenance of species diversity and population sizes. By definition, these values are not reliant on individuals visiting the site (so are not associated with measurable changes in behaviour). Brouwer et al (1997) provide a meta-analysis which attributes values to various ecological functions estimated from a large number of contingent valuation studies.

- *Hedonic Pricing.* Differences in property prices that can be attributed to aesthetic and amenity benefits of proximity to a wetland can provide a value for maintenance of biodiversity on the wetland site. This requires analysis of prices for otherwise similar properties that are located close to and distant from wetlands with a diversity of species.

- *Replacement costs.* The replacement cost of the biodiversity maintenance function is based on the costs of creation or renovation of an alternative. To provide a replacement, the alternative is required to provide similar habitats to the original site. Indeed, a possible management option would entail relocation of species to an alternative site. To have corresponding value, the alternative site is required to provide the same benefits. These are influenced by the location of the alternative site: the proximity to population centres, ease of access and availability of substitutes sites. There is also the question of 'authenticity': the original naturally occurring site may be preferred to an exact replica, thereby affecting amenity and non-use values. Valuation using the replacement cost is most straightforward for sites that predominantly provide the single function of biodiversity maintenance. For sites that provide multiple functions, the costs of replacement are attributed between the respective functions. Opportunity costs of the conversion of the alternative site and any externalities are also taken into account.

F3.2 Food web support

"*The support of food webs within and outside a wetland through the production of biomass and its' subsequent accumulation and export"*

The function of food web support comprises three processes:

- Biomass Production
- Biomass Import
- Biomass Export

Where on-site food web support = Production + Import – Export.

Of this food web support function, it is only the export of biodiversity from the wetland that is of economic value. While the export of biomass can result from physical means(the water course, overland flow, wind transport), or biological (associated with the movement of macro-invertebrates, birds, fish and mammals), it is only export by anthropogenic means that is of economic value.

Biomass Production

Biomass production occurs in any ecosystem, but it does not of itself provide economic value. Not all biomass will be equally beneficial in terms of the goods and services it provides, nor will all biomass necessarily be associated with any positive benefits in terms of satisfying human wants and needs.

Some work has been done to assess the productivity of wetlands in terms of energy flows (Farber and Costanza, 1987; Folke, 1991; Gosselink *et al.*, 1974), which is then converted into monetary terms according to the equivalent costs of deriving this energy from alternative sources (e.g. fossil fuels). The underlying assumption of such analysis is that all of the energy-potential stored in wetlands is economically useful (and equivalent to the alternative energy source). Since this will not be the case, and since there is no link between embodied energy and benefits derived from a wetland (apart from, perhaps, as an upper bound on consumptive use values: Costanza *et al.*, 1989), such 'energy analysis' cannot provide a realistic estimate of the value of a wetland. So long as all possible benefits to be derived from a wetland are identified, any value attributable to biomass production (and embodied energy) will be accounted for in the value estimates for other functions.

Biomass Import

The transfer of biomass **into** the system, or biomass import, is also not a valuable function in itself. Biomass import may contribute to functions within a wetland that are regarded as valuable. However, whether the biomass within a wetland derives from primary production or from import does not directly influence economic value. Even if these two different sources lead to distinct wetland functioning, it is the 'end result' of qualitative variations in ecosystem structure and biodiversity that determine the value of ecological functions. The value attributed to the maintenance of a given form of biodiversity will not alter according to from whence the biomass came.

Identifying the source of a wetland's biomass might become important if the valuation exercise is extended to systems beyond the wetland - perhaps to the catchment level - when functions performed within the wetland could be identified as relying upon the import of biomass from other ecosystems under scrutiny. Just as functions off-site can be linked to **export** of biomass from the wetland, functions within the wetland could be linked to the export of biomass by other ecosystems within the catchment.

Biomass export Through Harvesting (Non-Anthropogenic)

The biomass harvesting function is split into two distinct units for the purposes of valuation. The first, referring to **non-anthropogenic** or natural harvesting of biota, is considered here. The second, referring to **anthropogenic** export or harvesting, is considered in detail in the next section. The reason for splitting the function in this fashion for the purposes of assessing economic values is that deliberate harvesting suggests that benefits are being derived from the wetland's ecology, while a natural process of export need not necessarily be associated with any benefit to society.

The natural, or non-anthropogenic, harvesting of biomass is not of direct economic value - there is no demand for the physical process of biomass export. However, the biomass that is transferred to an alternative ecosystem by this process could be potentially valuable. Estimating the value involves assessing the benefits derived from the functions to which this biomass contributes in the alternative ecosystem, and attributing a portion of these benefits to the original wetland. As with other ecological functions that are associated with benefits derived from off-site locations (i.e. 'provision of habitat microsites'), quantifying possible economic values is likely to be a highly resource intensive exercise. It involves assessing the value of relevant goods and services provided by an off-site ecosystem and then attributing a portion of this value to the wetland's export function. Only where the wetland makes a significant contribution to off-site functions will such analysis be justified, which will involve repeating an assessment of the relevant functions being performed by an off-site ecosystem, determining the degree to which these are influenced by the transfer of biomass from the wetland, and applying appropriate valuation techniques to these functions.

Biomass export through harvesting (Anthropogenic)

The anthropogenic - as opposed to natural - export or harvest of biota from a wetland represents consumptive use value of the wetland. The assessment of the economic value of this function is simpler than for the other functions as the value of harvesting biomass is directly attributable to the wetland. All that is required is the identification of the impact on human welfare and the valuation techniques required.

Such export is associated with commercial exploitation of wetland resources, subsistence provision, or recreational use.

1. Commercial exploitation

Commercial value of wetland resources, for instance for fish, need, grass/hay or timber harvesting, can generally be assessed by analysis of market prices.

2. Subsistence use

   Subsistence value of wetlands is harder to estimate due to the fact that the products are not marketed. However, market prices may exist for the products, or market prices for alternative products or for the inputs to production (in particular labour), may act as surrogates for the price of wetland products.

3. Consumptive recreation

   Recreational value, such as fishing or hunting, is generally not associated with a functioning market. However, non-market valuation techniques such as the Travel Cost Method or the Contingent Valuation Method can be employed to assess these values.

An important aspect of valuation with regard to such consumptive uses is the degree to which extraction or exploitation is sustainable. While harvesting of a wetland's resources might provide a valid justification for its conservation, when compared with the opportunity costs of alternative uses of the land, the temporal scale over which these resources will continue to be available, and whether they may be over-exploited, will need to be considered. Where extraction is not sustainable, the impact that this could have on other functions and on future human generations should be assessed, as well as its effect on the economic value associated with the extracted resource. A pioneering study by Hammack and Brown (1974), considered the ability of wetlands to withstand pressure from recreational hunting, introducing this concept into their analysis via a 'biometric model'. Concepts such as sustainability constraints and safe minimum standards might usefully be applied.

Commercial Exploitation and Subsistence Use

Market analysis based on commercial exploitation of wetland products is based on observing prices that currently exist in markets for these products. However, market prices are often **not** a clear indication of the benefits derived by society from particular resources. For instance, taxes and subsidies, monopoly competition, and quantitative restrictions in markets for final goods or for inputs of labour and materials can distort prices. It may therefore be necessary to calculate 'shadow' or social prices for the relevant outputs and inputs, in order to derive **economic** value (as opposed to observed **financial** value) of the wetland function. Furthermore, if significant changes in supply or demand for resources are expected to result from wetland alteration, then any influence this might have on prices should also be taken into account. This is most likely to be a problem where large changes in resource availability are anticipated.

Consumptive Recreation

Consumptive recreational activities most often involve fishing and hunting. Since these activities are generally not associated with a functioning market, non-market valuation techniques can be employed. For instance, the Travel Cost Method, which uses expenditures on marketed goods involved in reaching a site, can be used to estimate the value attributable to recreation at that site. The Contingent Valuation Method can also be applied to elicit willingness to pay directly for recreational experience. These methods, and examples of their use, are illustrated in detail in the section 'Maintaining biodiversity.'

Annex G: Case Studies

<u>Case Study A: Flood Water Detention</u>

Example 1 The Introduction of a Flood Alleviation Scheme in the Broads and its Effect on Recreational Visitors: A Contingent Valuation Study

As a consequence of the ongoing and increasing risk of flooding within Broadland, in 1990 the National River Authority (NRA)[3] initiated a wide ranging study to develop an 'effective and cost-effective strategy to alleviate flooding in Broadland for the next 50 years' (Bateman et al., 1992, p. 31). The study consisted of five main components: hydraulic modelling; engineering; cost-benefit analysis (CBA); environmental assessment, and consultations. Given prior work on the market costs and benefits of flood alleviation schemes (Turner and Brooke, 1988), the principal task of the CBA was to estimate values for the non-market goods concerned. In particular, the estimation of the environmental and informal recreational values was seen as a central objective of this study. To address these values, in 1991 a CSERGE contingent valuation (CV) study was commissioned to assess the benefits of preserving the existing landscape, ecology and recreational characteristics of the area relative to their expected values in the absence of a Broadland-wide flood alleviation scheme (Bateman et al., 1992). The CSERGE study consisted of two surveys: (i) a postal survey of households across the UK designed to capture the values which non-users might hold for preservation of the present state of Broadland, and (ii) an investigation of the values held by users for the same scenario as elicited through an on-site survey. Details of these studies are presented below.

i) The 1991 study of non-users

Non-user values were estimated by means of a mail survey questionnaire sent to addresses throughout Great Britain selected so as to capture both socio-economic and distance decay effects upon stated WTP and thereby provide the basis for calculation of aggregate values. Full details of this study are provided in Bateman and Langford (1997).

The survey questionnaire was designed in accordance with the acclaimed 'Total Design Method' of Dillman (1978) and pre-tested through focus group and pilot exercises. Within the questionnaire was visual, map and textual information detailing the nature of Broadland, the flooding problems and flood defence options together with necessary details supporting a WTP question such as payment vehicle, payment time frame, etc. The survey achieved a typically modest response rate of some 31%, however initial

[3] National Rivers Authority (NRA), precursor to the UKs' Environment Agency with responsibility for the rivers.

analysis showed that this was heavily supported by past users of Broadland who represented well over one third of responses in each distance category. Although experience of visiting the Broads declines significantly with distance from the area ($p<0.0001$) it cannot therefore be claimed that the sample is nationally representative, rather it is biased towards past users and can be perhaps best characterised as being a sample of present non-users for comparison against results from the on-site survey of present users described subsequently.

When asked whether or not they agreed with the principle of incurring extra personal taxes to pay for flood defences in Broadland (the ‘payment principle’ question)[4] 166 respondents (53.5%) answered positively to the payment principle question.

Those respondents who accepted the payment principle were presented with an open-ended format valuation question asking them to state the maximum amount of extra taxes they would be WTP per annum to ensure the preservation of Broadland from the effects of increased flooding. Including, as zeros, those respondents who refused the payment principle (i.e. those who stated they were not willing to pay to prevent flooding), this question elicited a whole-sample mean WTP of £23.29 per annum (95% CI: £17.53 to £32.45)[5]. Table A1. decomposes these bids across a zonal distance variable, showing a marked decrease in mean WTP as distance from Broadland increases, and according to previous visitation experience, which shows that those who have previously visited Broadland express a substantially higher WTP than those who have not.

Table A1. Mean WTP per annum by distance zone and visitation experience (payment principle refusals included as zeros)

Distance zone	No. of respondents	Mean WTP (£ pa.)	Visit experience	No. of respondents	Mean WTP (£ pa.)
1(closest)	58	39.34	Holiday	118	27.86
2	66	27.67	Day trip	82	25.65
3	139	13.97	Never visited	110	12.29
4 (furthest)	47	14.72			
All	310	23.29	All	310	23.29

[4] The main aim of this question was to validate refusals as it was felt that respondents presented directly with a request to state how much they would pay might feel either intimidated about stating a zero amount (and consequently state false positives) or conversely, offended at the presumption of some positive WTP (which might lead to ‘protest’ behaviour; see Sagoff, 1988).

Analysis of WTP responses showed that bids were negatively related to distance from the Broads and positively related to income (or more precisely to that portion of disposable income which was spent annually upon recreational and environmental goods)[5].

Aggregation was conducted using three approaches:

(1) *Aggregation using sample mean WTP*. This approach was adopted in a recent high profile benefits assessment conducted by the Environment Agency (EA) of an application by Thames Water[6], to abstract water at Axford on the River Kennet, a low flow chalk stream in Southern England. Following existing guidelines (which are currently under revision) the EA multiplied a sample mean WTP by the entire population of the Thames Water catchment, yielding a very high estimate of preservation benefits which was rejected at a subsequent planning inquiry (which allowed the proposed abstraction to proceed). Following this procedure, the approach (1) multiplies the sample mean by the population of Great Britain;

(2) *Aggregation adjusting for distance zones*. Here a simple procedure was used, which could readily be adopted by policy analysts, wherein the mean WTP in each distance zone (detailed in Table A1) is used as the basis of aggregation. Multiplying the mean WTP by the population of each zone to capture a simple distance decay effect.

(3) *Aggregation by bid functions*. Here the consistent drivers of responses are recognised to have been the distance at which the respondent lives from Broadland and their socio-economic circumstances (specifically some measure of income), both in terms of the probability of replying to the questionnaire, of responding positively to the payment vehicle question and to the determination of the WTP amount.

Results from these various approaches to aggregation are detailed in Table A2.

Table A2: The present non-user's benefits of preserving the present condition of Broadland aggregated across Great Britain using various procedures (£ million/annum)

Aggregation approach	Untruncated	Truncated
(1) Aggregation using sample mean WTP	159.7	98.4
(2) Aggregation adjusting for distance zones	111.1	98.0
(3) Aggregation by bid functions:		
i. using distance zone and national income	27.3	25.3
ii. using county distance and regional income	25.4	24.0

[5] Further analyses showed included an assessment of WTP as a lump sum amount and the period of commitment for payments. Details are given in Bateman and Langford (1997).
[6] A private water company.

Examination of Table A2. shows that aggregate benefits estimates vary very considerably across the procedure used. Considering the untruncated sample we can see that the simplest approach (type (1) used by the EA in the Axford inquiry) results in an aggregate benefits estimate which is very much higher than that provided by even the crude incorporation of distance decay effects given under approach (2). However, benefits estimates reduce by an even wider margin when we move to the bid function approach (3) with the latter being around 1/6 of the initial estimates. Interestingly the bid function is relatively stable with respect to the scale of data used to calculate results although we would recommend the use of more detailed data wherever possible. Truncation effects are highly marked for the cruder aggregation approaches while the more detailed bid function approach yields estimates which are reassuringly stable.

In summary, the study of present non-users yields a consistent picture and provides the basis for some defensible estimates of aggregate benefits, which in turn yield an interesting commentary upon current practice. We now turn to consider the various on-site CV surveys of visitors to Broadland.

The 1991 study of users

From a decision making perspective, the 1991 on-site survey of those who directly use Broadland was primarily intended to estimate the benefits to this group of preventing saline flooding in the study area. However, the study also had an academic objective, namely to investigate the impact upon stated values of varying the way in which responses are elicited. Five WTP elicitation methods were investigated[7]:

(i) *Open-ended* (OE). Here the respondent is asked "How much are you willing to pay?" and is therefore free to state any amount;

(ii) *Single bound dichotomous choice* (1DC). Here respondents face a single question of the form "Are you willing to pay £X?" with the bid level X being varied across the sample;

(iii) *Double bound dichotomous choice* (2DC). Here those respondents previously asked the 1DC question are asked a supplementary dichotomous question on the basis of their prior response. Those who agreed to pay the 1DC bid face a higher 2DC amount while those refusing to pay the 1DC bid face a lower 2DC amount;

[7] Results from the open ended format are presented in Bateman et al., (1994) and contrasted with dichotomous choice and iterative bidding formats in Bateman et al., (1995). Analysis of the triple bounded format is given in Langford et al., (1996), Langford and Bateman (1999) and Bateman et al., (1999).

(iv) *Triple bound dichotomous choice* (3DC). This extends the previous procedure by adding a further question.

(v) *Iterative bidding* (IB). Here the bidding game formed by the various dichotomous choice questions is extended by a supplementary open-ended question asking respondents to state their maximum WTP. Such a procedure is appropriate to all respondents in the DC bidding game irrespective of whether they answer positively or negatively to individual DC questions.

The study itself generally conformed to the CV testing protocol laid down subsequently by the NOAA blue ribbon panel (Arrow et al., 1994). Survey design was extensively pre-tested with any changes to the questionnaire being re-tested over a total pilot sample of some 433 respondents. One of the many findings of this process was that a tax-based annual payment vehicle appeared optimal when assessed over a range of criteria (details in Bateman et al., 1993).

The finalised questionnaire was applied through on-site interviews with visitors at representative sites around Broadland with 2897 questionnaires being completed. This sample was composed of 846 interviewees being administered with the OE WTP questionnaire and the remaining 2051 facing in turn the 1DC, 2DC, 3DC and IB questions. Prior to any WTP question, respondents were presented with a 'payment principle' question. Negative responses to this question reduced sample sizes to 715 and 1811 respectively. Except where indicated, all those refusing the payment principle were treated as having zero WTP in calculating subsequent WTP measures. Table A3 details various measures of WTP estimated from these various elicitation methods.

Table A3: Measures of users WTP to preserve the Norfolk Broads from saline flooding obtained using various elicitation techniques

Elicitation Method	Mean WTP (£ p.a.)	Median WTP (£ p.a.)
OE	67.19	30
	(59.53-74.86)	(18-46)
1DC	144	144
	(75-261)	(75-261)
2DC	*	88
		(*)
3DC	*	94
		(*)
IB	74.91	25
	(69.27-80.55)	(19-35)

Notes: * = not estimated
Figures in parentheses are 95% confidence intervals

Table A3 indicates that there were strong elicitation effects at work in this case, although the direction of differences were as expected with OE estimates below those from DC approaches with IB values, the product of both methods, yielding intermediate results. There is considerable disagreement regarding the interpretation of such results. However, most commentators agree that OE responses are liable to represent more conservative estimates of underlying WTP. Given this, these results can be used as the basis for estimating lower values for the aggregate benefits to users of preserving the Norfolk Broads from saline flooding.

The Norfolk Broads CV study was conducted in answer to a real-world question regarding the funding of flood defences in Broadland. The study fed into a wider cost-benefit analysis which also examined the agricultural, property and infrastructure damage-avoided benefits of such defences. The benefit-cost ratio of the latter items was calculated at 0.98 (National Rivers Authority, 1992). However, even when the conservative OE measure of WTP for the recreational and environmental benefits of flood prevention is considered the benefit-cost ratio increases substantially to 1.94 (ibid.) indicating that the benefits of a flood alleviation strategy are almost twice the associated costs. These results including the findings from the CV study were submitted to the relevant Ministry of Agriculture, Food and Fisheries as part of an application for central government funding support for the proposed flood alleviation strategy. Following lengthy consideration of this application, in 1997 the Environment Agency announced that it had received conditional approval for a programme of 'bank strengthening and erosion protection' (Environment Agency, 1997).

Example 2 Sea Level Rise and the Threat to Broadland: a Cost Benefit Analysis

The physical vulnerability of the East Anglian coastline derives from large areas of low lying land, both immediately adjacent to the shoreline and inland, and stretches of soft erosive cliffs. Coastal defences, mainly hard engineering structures, built largely after the very destructive 1953 North Sea storm surge flood, and river flood embankment defences play a crucial role in the maintenance of the current shoreline, levels of economic activity and environmental resources in the immediate hinterland. The coastal erosion and flooding threat, enhanced as it may be by sea level rise associated with global climatic changes (Wigley and Raper, 1993), is directly related to the maintenance and ecological integrity of the Broads.

The policy options open in the face of this threat to the East Anglian coast include:

Retreat -	the abandonment of the land and structures in vulnerable areas and resettlement of population; this option can also include managed retreat linked to specific measures aimed at restoring or creating desirable habitat, landscape or amenity features;
Accommodation -	continued occupancy and use of vulnerable areas;
Protection -	continued full defence of vulnerable areas, especially population centres, economic activities and natural resources.

All of these options have direct consequences for the freshwater Broads themselves. The impact of accelerated sea level rise on the hazard zone depends greatly on how the coastline is managed. An economic approach to assessing which of the strategies is desirable for the overall coast and for each section of the coast, involves the quantification of costs and benefits. In this section, we restrict consideration to the two 'active' management response options: Accommodation and Protection, comparing these to the ‘Retreat’ option (full details of this and further analyses being presented in Turner et al., 1995). As in the case of the flood alleviation schemes considered previously, the cost benefit analysis is restricted to the economic benefit of protection to properties, agriculture and the indirect value of recreation and amenity in Broadland.

Cost-benefit analysis seeks to identify that strategy which maximises net present values. The benefits of protection and accommodation are the impacts avoided, taking the *Accommodate* option as the baseline scenario. These benefits therefore include the marginal value of the agricultural output, the other economic activity, and the amenity value of Broadland ‘saved’ through active management. The costs of the active strategies include hard and soft engineering defence maintenance; replacement cost of defences; cost of repairing breaches in defence, and necessary beach nourishment. Similar studies for developed and natural coastlines faced with the threat of climate change induced sea level rise include West and Dowlatabadi (1999), Yohe et al. (1995) and Yohe and Neumann (1997).

The total benefits of adopting either an *Accommodate* or *Protect* strategy are given by the size of the avoided damage costs, i.e. the change in costs compared to those incurred under the abandon defences (*Retreat*) option. The net benefits of each response strategy can be computed by subtracting the capital and maintenance defence system costs from the value of total benefits (damage costs avoided).

The results of the CSERGE cost benefit analysis are summarised in Table A4. The present value of flood and erosion defence costs for each response strategy is the accumulated cost of defence over the period 1990-2050, discounted at six percent[8]. There were no defence costs associated with the *Retreat* response strategy as this involves allowing natural erosion and flood processes to dominate the coastal zone. Benefits of the *Accommodate* and *Protect* strategies substantially exceeded those of the *Retreat* option and are shown as differences from the latter in Table A4.

Table A4. Costs and benefits of policy responses to sea level rise in East Anglia

	*Net present value over period 1990-2050 (£ million)***			
Sea Level Rise	0.20 m	0.40 m	0.60 m	0.80 m
Protection Costs (PC)				
Retreat	-	-	-	-
Accommodate	132	137	151	157
Protect	187	232	292	485
Flood and Erosion[1] Damage Costs (EC)				
Retreat	1333	1355	1405	1436
Accommodate	194	257	320	397
Protect	77	74	81	85
*Benefits (B)**				
Accommodate	1141	1108	1098	1058
Protect	1259	1284	1326	1352
Net benefits:(B-EC-PC)				
Accommodate	1009	971	947	**†901**
Protect	**†1072**	**†1052**	**†1034**	867

Based on Turner et al. (1995).

Notes:

* Benefits of defence relative to the do-nothing option ('Retreat'):

† Optimal strategy between accommodate and protect, given the sea level rise projection.

** NPV estimated at 6 percent discount rate.

[8] The appropriate UK Treasury specified discount rate.

1. Erosion accounts for around 20-30% of the total defence cost.

Although in the *Accommodate* response strategy, the physical scale of the defences was assumed to remain the same, there was still a gradual increase in defence costs with increasing values of sea level rise, due to an increased frequency of flooding events and consequent defence repair costs. In the case of the *Protection* response strategy, defence costs rise gradually, reflecting the need for progressively higher defences and increased volumes of beach nourishment required to counter sea level rise and enhanced erosion. The highest sea level rise projection (0.80 m) produces a significant increase in costs due to the necessity to employ relatively expensive engineering methods.

As far as the two active response strategies are concerned, the *Protection* response had the highest NPV indicating that this would be the preferred approach to coastal management for all sea level rise scenarios, with the exception of 0.8 m sea level rise. In this ‘extreme’ case, the high cost of maintaining a *Protection*-style defence line meant that an *Accommodation* response strategy becomes more desirable from an economic perspective. Due to the nature of the study area, the analysis was dominated by the flood hazard. In contrast to the overall results, when the erosion hazard alone was considered, the NPV was negative in almost all cases in the *Accommodation* and *Protection* response strategy simulations.

While physical parameters such as the assumed rate of sea level rise did have an effect on the results, socio-economic parameters were equally significant in this case study. Thus the extent and value of property at risk in the hazard zone dominated the damage cost calculations, although the environmental asset valuation is only a proxy variable and undoubtedly underestimates the full costs involved. Similarly the assumed discount rate had a major influence on the results. Sensitivity analysis of discount rates, from 3 to 9 percent, have the effect of doubling and halving the net present value of benefits and costs, from the base level of 6 percent rate. More importantly, they changed the rank order of the desirable policy options under some assumptions (Turner et al., 1995).

Case Study B: Ecosystem maintenance

***Sustainable Fen Management: An Economic Appraisal of Different Management Options (Ledoux* et al, 2000).**

The Broads represent the largest expanse of species-rich fen in lowland Britain. They contain over 250 plant species, from the nationally protected fen orchid, to the more abundant species such as ragged robin, a food source for the adult swallowtail, Britain's largest butterfly (Broads Authority, 1997). The fens are the first stage in the natural succession from open water to woodland. In former times, the fens were maintained by grazing animals, and were managed for reed and saw sedge for thatching, litter for cattle bedding and marsh hay. At present, however, only a few commercial uses, such as the harvest of reed for thatching, continue as viable industries. Without the economic stimulus of the traditional markets for other fen products, landowners cannot sustain the fen in an open condition, and many fens are left unmanaged, prone to natural succession and invasion by scrub. Without the removal of cut material, the deposition of dead vegetation and accumulation of organic matter lead to rising nutrient levels and drying out of the ground surface. The wetland will be gradually colonised with shrubs and trees and its value for wildlife habitat, nutrient removal, and flood storage will be reduced. Of the 5,000 hectares of semi-natural wetland remaining in the Broads, there are only 2000 ha of open fen, the rest having developed to form carr woodland. Since standard agricultural machinery cannot cope with the soft peaty ground and the thick fen vegetation, harvesting is often done by hand. Management of the fens is necessary for conservation purposes. This is, however, labour intensive, slow and costly, and also potentially ecologically damaging because the considerable amount of waste cut material is too costly to remove and often remains on site, resulting in further deterioration of reed beds and fens through nutrient enrichment.

Since 1996, new machinery and techniques have been developed for the large-scale sustainable management of wetland fens. A prototype harvester has been developed for harvesting and transporting cut material off site. This self-propelled, tracked vehicle is equipped with a cutter system, load bin, and discharge facility. It is able to cope with the most difficult wetland terrain, and supplies cut materials in an appropriate form for processing to create a variety of products, while minimising damage on sites that cannot support conventional equipment. Dykes are crossed using specially designed lightweight access ramps. The harvested material, once dried, is compacted into pellet form for storage and transport to outlets. Some dried material has been successfully utilised by a biofuel power station, and other uses and outlets are being investigated, such as animal feed, and supply to zoos.

The machine is expected to bring ecological benefits by allowing a regular management of the fens, without the soil compaction expected by standard agricultural machinery, and without the accumulation of waste material which occurs with hand cutting. Ecological data are currently being gathered to explore further the ecological impacts of the harvester and compare them with the impact of other management methods.

In this case study, the economic aspects of the different fen management options have been examined. The main objective was to assess the financial and economic costs and benefits associated with the three main options for fen management in the Broads: (1) light grazing, (2) cutting the vegetation using hand tools or (3) cutting the vegetation using a purpose built machine.

More specifically, the appraisal objective was a threefold output:

1) A *cost-effectiveness analysis* for these management options across the relevant marketing channels, *i.e.* from *in situ* cutting of the vegetation to storage, processing and transportation to various market outlets.

2) A *break-even analysis* for the management options, on the basis of existing prices, .

3) A financial *cost-benefit analysis,* which extends the cost-effectiveness analysis to the monetary economic benefits associated with each management option.

A more inclusive *economic* CBA (rather than a financial CBA) would incorporate the non-monetary ecological benefits by means of either benefits transfer or an original contingent valuation (CV) study.

As a first step, the relevant trajectory from *in situ* cutting to selling the cut material to various market outlets was identified. Site visits to see the harvest machine at work significantly increased knowledge of the operational procedures underlying fen harvesting.

For the first part of this trajectory, the actual harvesting of the fens, data collection sheets were prepared in close collaboration with the Broads Authority (BA), and tested in practice. First experiences with the use of these data collection sheets were discussed and the sheets were modified where necessary. Data for the harvest machine started to be collected in April 1998. Data were collected for the physical characteristics of the sites where the machine was used, working hours, machine costs and amount of material harvested. By the end of August, a machine that blows the cut material from the site to a lorry waiting off-site also became operational and was used at two different sites. Data about the costs of using the air blower were obtained for those two occasions.

The BA was contacted to examine the kind of information available for grazing. The BA also provided estimates of the total costs of hand cutting at the different sites where the harvest machine was used, based on the equivalent amount of manual work that would have to be carried out by local contractors. Site visits were also undertaken with the Norfolk Wildlife Trust (NWT) to see grazing experiments at Hickling Broad. Detailed information about the costs of grazing at three different experimental sites within the Broads was obtained from the NWT.

Scheme Costs

As pointed out at several meetings with the BA, the level of detail of the available data for alternative management regimes was a reason for concern. Regression and correlation analysis was used to examine the factors that influence fen management costs, both the total costs and the various cost components. Area size seemed to be an important factor. Although this conclusion was not clear-cut for its influence on total costs, it was clearly an important factor in direct harvesting costs, which confirms previous qualitative analysis: costs decrease as the area harvested increases. Accessibility was also a significant factor, mainly through preparation and maintenance costs.

Finally, vegetation variables were seen to be of varying significance. General vegetation type seemed to be a factor in explaining total and maintenance costs, but the high correlation coefficient with access made it difficult to form a clear conclusion. It was definitely a significant variable in direct harvesting costs. Specific vegetation layers, and litter in particular, seemed to have a significant role. Volume harvested per hectare was one of the most significant variables in explaining maintenance, harvesting, and total costs.

This analysis needs to be refined, as further information becomes available. The ultimate objective is to be able to explain the variability of costs according to site characteristics, in order to be able to predict costs on other sites. Ideally, the same analysis should be carried out for the alternative fen management options, in order to predict which option would be more effective at various types of sites, but the available data and the quality of these data were inadequate to allow such an analysis.

A number of cost categories were missing from the analysis, in particular the costs associated with using the air blower, transporting the cut material to a barn, storing the cut material and transporting the material from the barn to various market outlets. More insight is needed in what the real market options are, what the quality requirements for each market outlet are and what the prevailing prices in each of these markets are.

The BA foresaw carrying out the transportation of the machine and the cut material themselves. In that case, more detailed information is needed regarding costs of labour, fuel, insurance, etc. More detailed information about capital, depreciation (wear and tear of machine and other equipment) and insurance costs is needed to give a more complete picture of the total costs involved.

At the same time, the discrepancy between the detailed information for the harvester machine and the alternative management options will probably persist. This results in a rather unbalanced cost-effectiveness analysis, which includes extremely detailed information about the harvester machine and rough estimates for hand cutting and extensive grazing.

Cost-effectiveness analysis

Comparing the costs of the alternative fen management options was assessed at two levels: the costs of cutting the material and leaving it on site (i.e. without blowing, drying and transporting costs) and the net costs including transport off site and selling the cut material. Three approaches were assessed: using the harvester, hand cutting through local contractors and extensive grazing. The results showed that cattle grazing is the least cost option, however, this can only take place in specific sites (i.e. where upland areas are available for grazing during the winter when the fen is flooded), otherwise the harvester is the least cost option.

Break-even analysis

All sites assessed in this study showed a net cost as the costs per ton of transporting the cut material to the barn, drying, and transport to market outlets are higher than the £100/t assumed market price (based on price for animal feed). The analysis showed a wide range of variability in the break-even price for each site (£350-£900/t) these were generally a lot higher than the forecasted £100/t.

Cost-benefit analysis

A cost-benefit analysis of the harvester project was done over a period of 10 years (roughly comparable to the period of life of agricultural machinery, Nix (1987). The total annualised costs and benefits are given in Table B-1. The harvester schemes produces an annual net cost of £120868, which discounted over 10 years at a 3% discount rate gives a net present value of -£1031029. Therefore, on narrow economic grounds, the project should not go ahead, but this includes only the financial benefits and did not include the broader economic benefits such as recreation and existence values. Revenues from the Fen tier ESA (Environmental Sensitive Area) payments can be seen as a proxy of these benefits, but these are probably a gross underestimation of the value, as they are based on the costs incurred in managing these habitats. Furthermore, they are designed to be an incentive to management, and cover only a fraction of the total costs of management.

The financial CBA was intended ultimately to result in a more inclusive economic CBA where the ecological impacts are also accounted for. In order to progress in this direction, further information from the BA would be needed about the ecological impact assessment carried out.

Table B-1 Total annualised costs and benefits (Calculated for 40 ha of Fen)

Costs	£/ha	£(40 ha)	Benefits	£/ha	£(40 ha)
Total preparation Costs		**12730**	**Directive extractive benefits –** Market revenues	**630**	**25087**
Labour	304	12160	**Fen Tier (ESA payment)**	**120**	**4800**
Capital depreciation		570			
Total Harvesting Costs		**35439.7**			
Labour (harvest & maint.)	515	20600			
Fuel	17.5	700			
Capital depreciation		14139.7			
Total Blowing costs		**37577.4**			
Labour (blowing & maint.)	588	23520			
Fuel	36	1440			
Capital depreciation		12617.4			
Total Drying Costs		**27580.5**			
Labour	433	17320			
Fuel	7.6	304			
Electricity	172	6880			
Capital depreciation		3076.5			
Total Transportation Costs		**24343**			
Transport of harvester					
Labour	34.5	1380			
Fuel	1.2	48			
Transport of materials to barns					
Labour	272	10880			
Fuel	9	360			
Transport to markets	145	5800			
Capital depreciation		5875			
Total general annual costs		**12950**			
Health and safety		300			
Tools		150			
Oils and Lubricants		1000			
Repairs and maintenance		7500			
Lease buildings		4000			
Total one-off costs		**134.4**			
Training		88.1			
Electrical Work		46.3			
Total		**150755**			**29887**

Knowing the ecological impacts of using the harvester machine and hand cutting or extensive grazing at the different sites, would mean that the assumption that the same results are achieved irrespective of the management option chosen, can be relaxed.

Finally, there is an ongoing Geographical Information System (GIS) analysis, which seeks to explain and map the variability of costs according to site characteristics, in order to be able to predict costs on other sites. A GIS already maps vegetation types in various areas in the Broads. The objective was to use this existing system to include the collected costs and benefits of different fen management regimes at a number of specific sites. Including the transportation network would also enable the calculation of transportation costs to potential market outlets. This would allow the results from the cost-benefit analysis to be used to determine the viability of other presently unharvested sites, based on vegetation characteristics of the sites, their accessibility to markets, and their suitability based on optimal block sizes. Overlapping these economic maps with the BA ecological objectives should prove to be a very useful tool in strategic fen management plans.

Case Study C: Wetlands Creation/Managed Re-alignment

Example 1 Knottingly Flood Allevation Scheme (ENV Agency, 2003)

Following limited flooding in November 2000, a Preliminary Strategic Review (PSR) was conducted in May 2001 for the Lower Aire Catchment which identified problems at Knottingley and specifically in the Gander Haven area of the town. Knottingley is located on the south bank of the River Aire approximately 20 km to the east of Leeds. Much of the town is low lying and is protected from flooding from the Aire by artificial raised defences (floodbanks). The objective of the Knottingley Flood Alleviation scheme is to reduce the risk of flooding to Knottingley in a manner that is economically, environmentally and technically feasible. Following a feasibility study which identified causes and potential solutions to the flooding issue, a cost-benefit analysis (CBA) was undertaken to determine the preferred solution. This process included an environmental impact assessment of the preferred option in conjunction with a consultation with all the local residents and other stakeholders including the following bodies: Environment Agency; English Nature; English Heritage; Yorkshire Wildlife Trust; Countryside Agency; DEFRA; City of Wakefield MDC and the West Yorkshire Archaeology Service.

The Flooding Problem

The flood defences in the Lower Aire Catchment were constructed to a 1:100 year standard. In the Gander Haven area, the original defences comprised of a low earth flood bank with a reinforced concrete post and panel arrangement on the crest of the steeply sloped riverbank adjacent to Gander Haven Farm. During the flooding of November 2000, several water flows were observed at a number of locations suggesting that flow paths exist through and under the riverbank. The site inspections revealed that the structural integrity of the flood banks do not provide a sufficient level of protection, with water seepage being the key concern.

The weak defences are a consequence of a combination of over-steepness combined with the development of seepage flow paths through the riverbank. Upstream and adjacent to Gander Haven Farm, the floodbanks are constructed from poor quality material comprising of topsoil underlain by sandy clay, through which water seeps through it when the river level is high. Downstream of the Farm, while the bank is well formed: the banks have low gradients, a wide crest and are clay cored, a silt layer underlies the floodbanks which provides a seepage path and potential plane of failure.

After the flooding in November 2000, emergency works were undertaken to reinforce the original defences. A concrete wall was constructed immediately in front of, and to the same height as, the existing concrete post and panel fence, extending from ‘The Flatts’ at Knottingley for approximately 250m until it ties into high ground downstream of the farm property. The flooding at Gander Haven farm can be attributed to seepage as opposed to overtopping as the existing riverbanks are constructed to a 100-year minimum defence standard level of protection in this vicinity.

In order to retain the existing level of flood defence, the Knottingley Flood Alleviation scheme aims to sort out the seepage problem at Gander Haven Farm. It is considered to be a standalone scheme, having no detrimental impact upstream or downstream of Knottingley. The flood defences for in the Lower Aire are constructed to a 1:100 year standard - constructing defences at Gander Haven at a higher standard than 1:100 level would put cells adjacent to Gander Haven (Ferrybridge and Knottingley) at increased risk. Even if defences for Gander Haven were constructed to a higher standard, the area would still be at risk of flooding "round the back" for example from the Aire and Calder navigation, as much of the land is very low lying. The effects of climate change will be addressed by improvements to the surrounding washlands; therefore the height of the proposed defences and the existing defence level will remain unchanged.

Options Selection

A comprehensive assessment of the mechanisms and consequences of flooding identified the following options to provide flood defences to a 1 in 100 year standard of protection by stabilising the river banks and preventing water seepage. A true “do nothing” scenario which provides less than 1 in 100 year level of protection is also considered which acts as a reference point to compare the costs and benefits of flood protection. A detailed description of each of the options follows.

Option 1 – Do Nothing

- The economic baseline for the studies and assumes no further work or expenditure
- Less than 1 in 100 year level of protection: risk of inundation to 244 properties.

Option 2 – Do Minimum

- Retains current level of protection - preventing the progressive failure of the existing defences.
- Riverbank stabilised through installing a toe revetment (additional rock required every 5 years).
- Seepage prevented by driving steel sheet piles downstream (6m deep and 600m long).

Option 3 – Install Bentonite Retaining Wall & Sheet Piled Cut-Off

- Riverbank stabilised – removal of existing concrete floodwall, installation of a 250 m long, 12 m deep cut-off wall made from cement-bentonite slurry on riverbanks.
    - Requires purchase and demolition of Gander Haven Farm.
    - Risk of slurry entering the river via spillage during construction or via seepage through the embankment.
- Seepage downstream prevented by driving steel sheet piles (6m deep and 600m long).
- Life of existing riverbank extended by installing a toe revetment system (replaced after 25 years).

Option 4 – Install Steel Sheet Piled Wall

- Riverbank stabilised by removing the existing concrete floodwall and installing of a 250 meter steel sheet piled wall driven to a minimum of 12 metres below ground level to tie in with suitable founding stratum.
    - Requires the purchase and demolition of Gander Haven Farm.
- Seepage downstream prevented by driving steel sheet piles (6m deep and 600m long).
- Life of existing riverbank extended by installing a toe revetment system (replaced after 25 years).

Option 5 – Install Secondary Defence and Sheet Piled Wall

- Earth embankment constructed immediately upstream of the Farm to retain the seepage.
    - Creates a small wetland area - too small for washland storage.
    - Excess water removed from enclosure after the river levels have subsided by a pumped outfall or gravity drain.
- Seepage downstream prevented by driving steel sheet piles (6m deep and 600m long).
    - Requires the purchase and demolition of Gander Haven Farm.
- Life of existing riverbank extended by installing a toe revetment system (replaced after 25 years).

Option 6 – Managed Realignment (i) – Locally Realign Primary Defence

- Localised retreat of the primary defence line to create a more stable slope.
    - Requires the purchase and demolition of Gander Haven Farm.
- Seepage prevented by driving steel sheet piles downstream (6m deep and 600m).

Option 7 – Managed Realignment (ii) – Large Scale Retreat

- Large-scale retreat of the primary defence line to create a more stable slope.
    - Requires the purchase and demolition of Gander Haven Farm, in addition to EA owned land downstream.
- Creation of a wetland area of approximately 6 hectares.

Option 8 – Managed Realignment (iii) – Retain Building Plot

- Large-scale retreat of the primary defence line to create a more stable slope.
    - Requires the purchase and demolition of Gander Haven Farm.
    - Uses EA owned land downstream of Gander Haven farm.
- Land with planning permission is defended upstream - steel sheet piles driven into riverbanks.

A summary of the environmental impacts of the options considered is presented in Table C1.

Table C1 – Summary of Environmental Impact of Options

Option	Negative Environmental Impact	Positive Environmental Impact
1) Do Nothing –	Risk of flooding. Continued erosion of flood banks and loss of foreshore. Unsustainable.	None.
2) Do Minimum	Damage to banks via piling and stone revetment. Unsustainable.	Foreshore maintained
3) Install Cut off wall	Damage to banks. Pollution potential. Unsustainable.	Foreshore maintained.
4) Sheet steel piled wall	Damage to banks. Unsustainable..	Foreshore maintained.
5) Secondary defence & sheet piled wall	Damage to banks. Loss of semi improved pasture.	V. small wetland created. New planting along new defence. Partially sustainable.
6) Managed realignment – Locally realign + sheet piling	Damage to banks. Loss of semi improved pasture.	Small area of wetland created. New planting along new defence Partially sustainable.
7) Managed realignment Large scale retreat	Loss of semi improved pasture.	New wetland habitat created for other species of fauna, including water voles, birds, etc. Possible refuge for Otters. Wetland will comprise 6 hectares of reedbed, wet grassland and scrub. Additional planting included to enhance the surrounding area. Will provides a more diverse landscape for the community. Increased opportunities for recreation and landscape. Meets a number of high level environmental targets and also providing an amenity to the local community. Allows natural river bank erosion within the wetland area. Sustainable option.
8) Managed Realignment Retain building plot	Damage to banks As above but less sustainable	Foreshore partially maintained. As above but smaller area of new habitat.

Costs of Options

Scheme Cost Estimates

- Produced with Contractors and designer –budget estimates based on conceptual designs
- Capital costs derived on the basis of unit rates listed in the CESMM3 Price Database (1999-2000).
- Costs comprise of the initial design fees and construction costs together with long term reconstruction and maintenance costs.

Benefits of Options

Scheme benefits Estimates - following DEFRA Flood and Coastal Defence Project Appraisal Guidance Notes - Economic Appraisal (FCDPAG3, December 1999).

- Scheme benefits calculated on the basis of flood damages.

- Do Nothing' Option damages are calculated assuming an abandonment of the defences and a complete 'write-off' of all properties affected once a failure has occurred.
- Benefits of Managed Realignment:
  - Enhanced habitat for protected species, including nationally and internationally rare birds, mammals such as Water Voles and other species of interest.
  - Proximity of Knottingley and areas of new housing nearby – potential users of the site, e.g. dog walking, fishing. The nearby Fairburn Ings RSPB Reserve receives approximately 55,000 visitors a year.
  - and the site will be an educational resource -
  - Support of national environmental policy by contributing to new national Biodiversity Action Plan habitat;
  - Meeting DEFRA high level Target 9 – development of reed beds.
  - For the biodiversity of the wetlands created in options 7 & 8, a value of £132,500 was used. For a number of reasons detailed below in "valuing biodiversity" this value is incorrect.

- All of the benefits relate to damage avoidance and no allowance has been made for the following:
  - Loss of Life.
  - Anxiety and Stress of Residents during prolonged rainfall.
  - Difficulties in Obtaining Insurance.
  - Traffic disruption.
  - Re-housing residents.
  - Emergency services response.

Valuing Biodiversity

The figure used in this report to denote the value of biodiversity for created wetland area contains a number of errors. The value of £132,500 is comprised of two components:

1) English Nature report (no 406, 2001) reported the capitalised values per hectare for the Total Economic Value (TEV) of wetlands developed in a RPA report (1998) which were further refined in the appraisal process to give a capitalised value of £20,000 per hectare for the land at Gander Haven Farm based on the good conservation quality land, predicting that the site would contain 2 national BAP habitats and 6 local BAP species.
   Thus, the value of the 6 hectares of wetland created under this scheme is **£120,000.**

2) The following payments from the countryside stewardship scheme (CSS):

Reedbed creation:	£100/ha/yr
Access:	£150/ha/yr
Educational Access:	£500/yr
Total:	**£1,250/yr**

Total over 10 year period: **£12,500.**

Giving a value for the created wetland of **£132,500 (**£120,000 + £12,500)

Sources of error:

1) **Discounting:** in calculating the proxy value of the habitat through payments made through the CSS, the payments which occur over the 10 year period are not discounted. Despite the stated use of a 6% discount rate. Plus a 10 year time period is used for these payments despite the 50 year design life. Thus, £1,250 discounted over a 50 year timespan would be: £20,952.

2) **Double-counting:** the total value of £132,500 for environmental benefits reported in this study is based on a total economic value for wetlands AND a proxy value of the environmental benefits (CSS payments).

Thus, the environmental benefits have been included twice in the CBA. Either the total economic value for wetlands derived from the RPA report (£120,000) or the proxy value from the Countryside Stewardship Scheme (£20.952) should have been used.

Choice of Preferred Option

- Based on an over-estimated value of biodiversity – the highest value was £120,000
- Assumed design life of fifty years (present values discounted at 6%, sensitivity testing using a 3.5% discount rate and a doubling of the benefits).
- A residual risk contingency of 95% was allowed for. The risks identified at this appraisal stage tended to be very specific in nature, and related to uncertainties in the design detail. The sum identified is included in the final cost estimate as a contingency.
- All derived cost estimates were updated to 2002 figures on the basis of Retail Price Indices (RPI's) provided by the UK Office of National Statistics.
- Average benefit/cost ratio includes Environment Agency staff costs and risk contingency.

The results of the benefit-cost analysis are shown in Table C-2. The preferred option was shown to be the large scale retreat (Option 7) with a benefit/cost ratio of 5.02, based on the use of the overestimated value for biodiversity. However, an assessment of the figures in C-2 shows that overestimates the benefits of biodiversity has limited impact on the overall result. Option 7 comprises the provision of a 1 in 100 year standard defence to the area at risk of flooding on the south bank and the construction of a new realigned primary defence, and would require the demolition of Gander Haven Farm. Implementation of this enhanced Managed Realignment option would provide a solution that addresses holistic reach management, i.e. the option would relieve the pressure on the river bend immediately downstream of Gander Haven Farm. Therefore, implementation of this option would reduce the future level of maintenance (stoning) required at the river bend. This option would also ensure that the existing defence levels would not exceed the original arrangement, therefore avoiding any detrimental effects to the existing washland system. This option also conforms to the Sustainable Construction Policy.

Table C2 –Benefit Cost Analysis for Alternative Options

Options	**A Present Value Expenditure (£)**	**B Present Value Direct Flood Damages (£)**	**C Present Value Benefit Achieved by Scheme (£)**	**D Average Benefit/ Cost Ratio ***	**E Average Benefit/ Cost Ratio ****
1 – Do Nothing	0	9,817,571	0	-	-
2 – Do Minimum	1,164,624	9,817,571	0	-	-
3 – Install Bentonite Retaining Wall & Sheet Piled Cut-Off	2,462,196	4,184,616	5,632,955	2.29	1.97
4 – Install Sheet Piled Wall	2,475,403	4,184,616	5,632,955	2.28	1.96
5 – Install Secondary Defence & Sheet Piled Cut-Off	2,159,365	4,184,616	5,632,955	2.61	2.20
6 – Locally Realign Primary Defence & Install Sheet Piled Cut-Off	1,521,532	940,052	8,877,519	5.83	4.63
7 – Large Scale Retreat	**1,399,197**	**940,052**	**9,010,019**	**6.44**	**5.02**
8 – Retain Building Plot	2,983,199	940,052	9,010,019	3.02	2.67

* Excluding Environment Agency staff costs and risk contingency
** Including Environment Agency staff costs and risk contingency

Incremental benefit cost ratios have not been calculated as part of the economic appraisal, as all options (except Option 1 the baseline) will provide the same level of protection (100-year defence level).

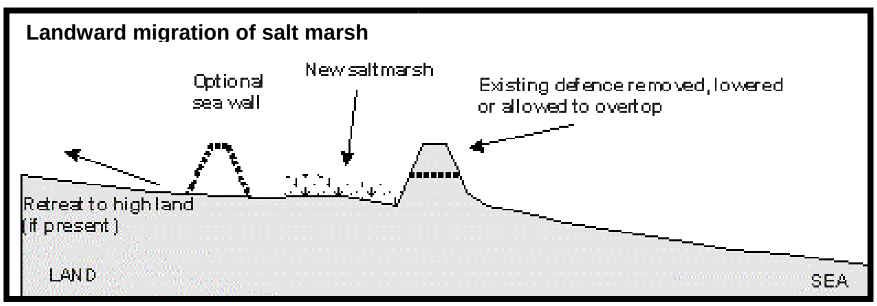
However, the economically preferred option does not change under the sensitivity analysis, highlighting the robustness of the recommendations made.

Example 2. Managed Realignment in the Humber.

(Further detail can be found in Burgess *et al.* (Forthcoming))

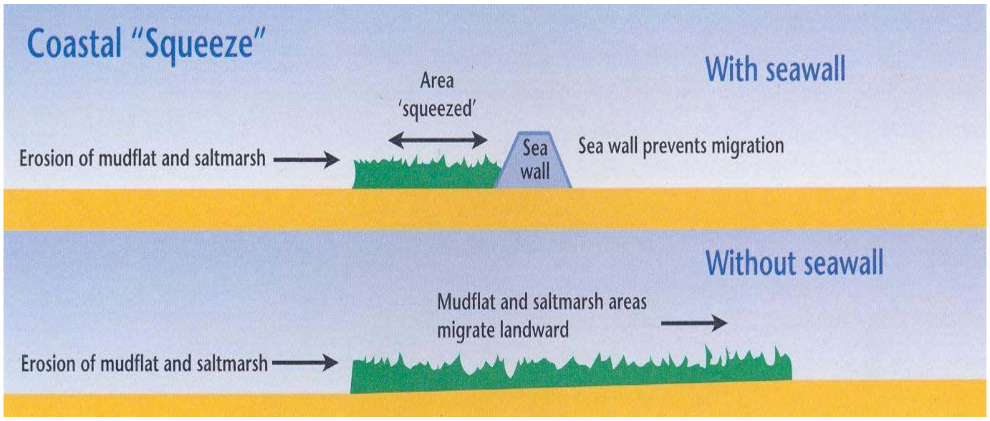
With climate change leading to rising sea levels and increased storminess and many of the existing defences reaching the end of their design life and in need of repair or replacing, the sea defences and coastal protection strategies of the UK are currently under review. The traditional method of flood management has been to construct structural or 'hard' defences, such as sea walls to protect land, however a re-orientation in thinking about response strategies is leading to an abandonment of "coastal armouring" (i.e. the building and maintenance of hard engineering structures and works) in favour of a mixed approach which will include substantial elements of more flexible "soft engineering" measures such as set back or managed realignment The term 'managed realignment', also referred to as 'managed retreat' or 'coastal setback' (Reed et al., 1999), involves deliberately breaching engineered defences to allow the coastline to recede to a new line of defence further inland. See figure C 1.

Figure C-1 Managed Realignment



Although the contemporary belief was that hard defence was best practice, it is now recognised that these are unsustainable both from an environmental and economic perspective given recent understanding of coastal dynamics (Crooks *et al.,* 2001). Not only do these hard defences provide a false sense of security and encourage development immediately behind the defences, they show little regard for natural processes (Crooks *et al*, 2001). Natural responses to rising sea levels are prevented by sea walls and other hard flood defences; for example, the erosional response in which the intertidal boundary migrates landward is prevented by flood embankments which results in 'coastal squeeze' (Pethick, 2001). This is shown in figure C-2, this issue of eroding sea-defence lines is exacerbated by climate change (Möller et al., 2001), with the potential for intertidal habitats to be lost completely (O'Riordan et al., 2000).

Figure C-2. Coastal Squeeze (Environment Agency, 1998).



Evidence of where adaptation to sea level rise has been prevented by hard defences can be seen along the Essex coastline, UK. Medieval to 19th century embankments have led to the loss of 40,000ha of saltmarsh. These embankments have restricted the landward migration of intertidal habitat and have resulted in high-level saltmarsh and natural transition zones between the sea and the mainland becoming very restricted (Dixon et al., 1998).

Managed realignment schemes generally aim to realign defences in a manner that will not only reduce the length of defence required, but will also increase the overall area of intertidal habitat. This is partly to create intertidal habitat as a means to comply with the Habitats Directive and also because it has recently been recognised that the intertidal zone may act as a natural sea defence by absorbing energy and water reducing the potential for a flood to be hazardous (O'Riordan et al., 2000).

Managed Realignment in the Humber

The macro-tidal Humber estuary is one of the largest in the UK, with a maximum tidal length of 147 km from Cromwell Weir on the Trent to the Humber's mouth, and maximum width of 15 km, comparable with the Thames and Severn Estuaries (Andrews et al, 2000). The area surrounding the Humber Estuary (generally referred to as Humberside) is mainly high quality agricultural land, with many thousands of hectares reclaimed from the estuary over the last few centuries. As a result, it is estimated that over a third of a million people now live on areas of land below high spring tide level. Approximately 570 km2 of land would be flooded if the present extensive coastal defences were removed.

Andrews et al. (2000) have calculated the intertidal area of the Humber to be 111km^2 (11,100ha), of which 90% is mudflats and sandflats and the remainder, saltmarsh (Winn et al., 2003). This intertidal habitat is particularly important in terms of bird species and the Humber is recognised internationally for its breeding, passage and wintering birds. The entire estuary has been proposed as a marine 'Special Area of Conservation' (SAC) (Ledoux et al., 2002) while the Humber Flats are designated a 'Special Protection Area'

(SPA), 'Site of Special Scientific Interest' and Ramsar site (Conlan and Rudd, 2000). Under the Habitats Directive any habitat loss from SAC and SPA areas is required to be compensated for. More than 90% of the intertidal area and sediment accumulation capacity of the Humber estuary has been lost over the last 300 years (Jickells et al., 2000), with protected areas becoming threatened.

Investigations using GIS, have identified suitable areas for realignment within the Humber based on the following five key issues (Coombes, 2003).

- Area below the High Spring Tide Level (maximum area of potential intertidal habitat).
- Present Land Use - undeveloped land more suitable for conversion (physical ease (Reed et al., 1999) and its economic value). However, Sites of Special Scientific Interest (SSSI), Special Areas of Conservation (SAC) and other similarly protected areas may not be considered suitable.
- Infrastructure (transport network, including roads, railway lines and canals)
- Historical Context.
- Spatial Context of the Areas:

    - SIZE: realignment is not cost-effective for areas under 5ha. (Pilcher et al. 2002).
    - SHAPE: - trade-off between a wide intertidal area to maximise benefits and length of realigned defences to protect the surrounding land (Pilcher et al., 2002).
    - ELEVATION: higher ground can be used as a natural defence to absorb wave energy to minimise defences required and reduce maintenance costs of the realigned defences (O'Riordan et al., 2000).
    - PROXIMITY TO EXISTING INTERTIDAL HABITATS: to facilitate the movement of species between habitats (Begon et al., 1996).

To help reduce uncertainty and aid decision-making, managed realignment was assessed for five distinct scenarios: Hold-the-line, Business-as-usual, Policy Targets, Deep Green and Extended Deep Green. The Business-as-usual and Policy Targets scenarios consider realignment areas within the estuary, whereas the Deep Green and Extended Deep Green scenario additionally consider realignment in the tidal reaches of the Rivers Ouse and Trent. These areas are shown in Figure C-3, with detailed information of each scenario given in Table C-4.

Figure C-3. Areas suitable for managed re-alignment in the Humber

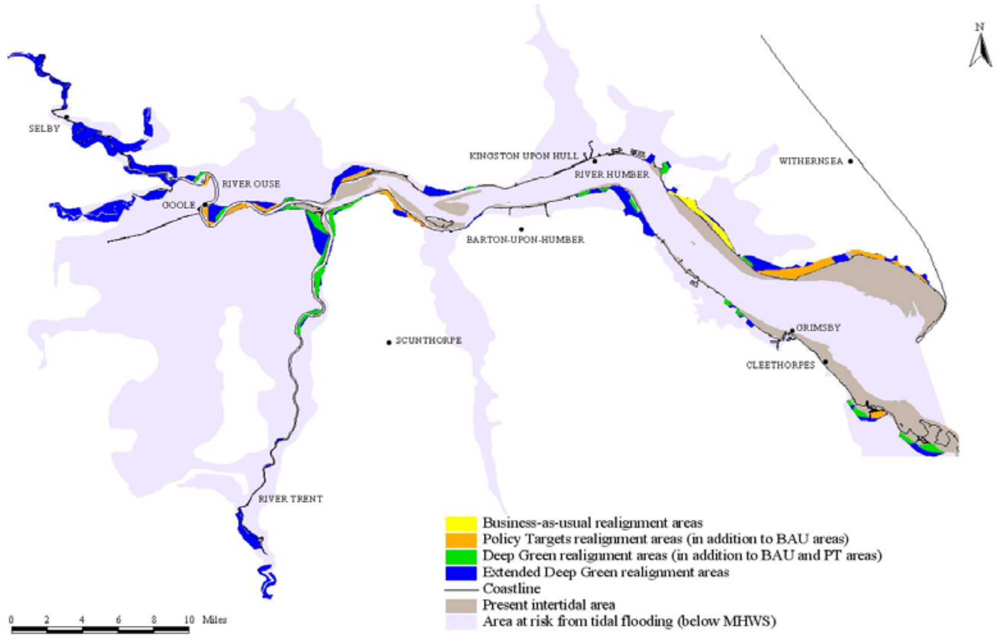


Table C-4: Details of areas suitable for realignment.

a) Within the Humber Estuary.

Scenario	Length of defences before realignment (km)	Length of hard defences after realignment (km)*	Length of realigned defences (km)	Length of unsatisfactory defences after realignment (km)	Intertidal habitat created by realignment (ha)**	Carbon stored (tonnes/yr) (Andrews et al, 2000)**
Hold-the-line	227.4	227.4	0.0	64.6	0.0	0
Business-as-usual	227.4	225.9	7.0	61.9	300.2	660.6
Policy Targets	227.4	214.5	30.8	42.2	1,320.9	2908.8

b) Within the Tidal Reaches of the Humber.

Scenario	Length of defences before realignment (km)	Length of hard defences after realignment (km)*	Length of realigned defences (km)	Length of unsatisfactory defences after realignment (km)	Intertidal habitat created by realignment (ha)**	Carbon stored (tonnes/yr) (Andrews et al, 2000)**
Hold-the-line	405.3	405.3	0.0	64.6	0.0	0
Business-as-usual	405.3	396.8	7.0	61.9	300.2	660.6
Policy Targets	405.3	361.6	30.8	42.2	1,320.9	2908.8
Deep Green	405.3	318.2	69.0	38.2	2,332.4	5135.1
Extended Deep Green	405.3	284.5	102.7	34.0	7,493.6	16501.8

* The process of realigning defences can involve the maintaining of existing defences and potentially the creation of new sea walls (see Box 5) therefore the length of defences after realignment is greater than the length before minus the length to be realigned.

**Due to uncertainty over the loss of intertidal habitat due to coastal squeeze over the next 50 years, it is assumed that no further coastal squeeze takes place. Therefore, the Hold-the-line scenario as the baseline scenario assumes no loss of intertidal habitat and no carbon sequestration, and the habitat creation and sequestration in the other scenarios are relative to base.

Project Assessment of the Managed Realignment Schemes

Cost Benefit Analysis (CBA) was used to determine the economic efficiency (whether benefits exceed costs) of each of the managed realignment scenarios.

Costs of Realigning:

- Capital Costs - the costs of realigning the defences
- Cost of land – The value of the land to be converted into intertidal habitat.
- Replacement Costs - The costs of replacing unsatisfactory defences.
- Maintenance Costs - Both hard and realigned defences must be maintained at a satisfactory level.
-

Benefits of Realigning

- Creation of intertidal habitat - a composite environmental benefit value of habitat creation.
- Carbon Sequestration – storage of carbon
-

Choice of Preferred Option

(Following: Flood and Coastal Defence Project Appraisal Guidance: Economic Appraisal: Supplementary Note to Operating Authorities. REVISIONS TO ECONOMIC APPRAISAL PROCEDURES ARISING FROM THE NEW HM TREASURY "GREEN BOOK" DEFRA (2003)).

- Assumed design life of 100 years.
- Present values discounted using a declining discount rate

Period of Years	0-30	31-75	76-125	126-200	201-300	300+
Discount Rate	3.5%	3.0%	2.5%	2.0%	1.5%	1.0%

- All estimates were updated to 2001- 2002 figures using GDP deflators.

Item	**Value after adjustment to year 2001-2002**
Capital costs of realignment (realigning defences)	£811,893/km
Opportunity costs (Grade 1 and 2 land)	£2,110/ha
Opportunity costs (Grade 3 land)	£2,382/ha
Maintenance costs of realigned defences	£1,239/km/yr
Maintenance of non-realigned defences (max estimate)	£5,127/km/yr
Replacement costs	£618,000/km
General habitat creation benefits	£574/ha/yr
Carbon sequestration benefits	£7.18/tonne CO_2e

The results of the benefit-cost analysis are shown in Table C-5. This showed that all the scenarios over a 100 year life span showed positive benefits, however greater benefits result from the schemes with greater levels of managed realignment. Further sensitivity analysis showed this result to be robust.

Table C-5. CBAs for all the scenarios compared to the HTL scenario.

	Scenario:	100 years	Compared to HTL
Humber Estuary	**Hold the line**	-119695239.4	
	Business as usual	-115922399.5	**3,772,839.903**
	Policy Targets	-94371704.89	**25,323,534.48**
Tidal Extent	**Hold the line**	-182121725.4	
	Business as usual	-178348885.5	**3,772,839.903**
	Policy Targets	-156798190.9	**25,323,534.48**
	Deep Green	-154481486.1	**27,640,239.31**
	Extended Deep Green	-82493880.53	**99,627,844.89**

Case Study D: Wetlands Restoration

The restoration of the Culm Grasslands: a scientifically important wetland ecosystem

The Culm Grasslands are semi-natural grasslands in South West England which have resulted from traditional low-intensity farming practises. These grasslands are a scientifically important habitat home to many endangered species including the Marsh Fritillary Butterfly and the Curlew. Approximately seventy-five percent of the grasslands are under some form of management agreement (Hocking and McCartney, 1999; Saunders, 2003). However, the grasslands are highly fragmented and are only a small remnant of there former occupation within Devon and Cornwall; an estimated 92% of the grassland have been lost since 1905 (English Nature, 1991). Agricultural improvement has frequently been cited as the principle driving force behind these massive losses (i.e. Hocking & McCartney, 1999; Aitchinson & Ashby, 2000; Devon Biodiversity Partnership, 1998).

The remaining grassland is therefore vulnerable to the effects of adjacent land use and neglect (Saunders, 2003). The main threat to the conservation of the Culm Grasslands not under management agreements is land use change, either through further agricultural intensification, or more probably, through neglect due to changes in CAP regime and the diversification of farming, often towards tourism (England Rural Development Programme, 2000). However, the longer term ecological viability of the majority of the Culm Grassland sites (whether under management agreements or not) is seriously in doubt given their small size and patchwork pattern (Saunders et al., 1991; Debinski & Holt, 2000; Lahti, 2001).

In the face of the continued losses of the Culm Grassland there are a number of future policy scenarios that could be undertaken; these include:

Business As Usual (BAU) - Continued losses to the remaining areas of Culm Grasslands. This is a result of further land use change due to reform of the CAP as well as decreased ecological viability of the remaining sites. The long term viability of the Culm Grasslands is seriously in doubt under the BAU scenario.

Policy Targets (PT) - As a consequence of the ongoing risk to the future integrity of the Culm Grasslands, in 2003 the Devon Wildlife Trust set a target to increase the grasslands by 10% over the next ten years (Saunders, 2003). This target is implemented under the PT scenario. There will not only be improvements in the quality and longevity of the grasslands, but this can be coupled to sustainable rural development that will draw on and enhance the environmental and cultural assets of the Culm.

Deep Green (DG) - protection of the Culm Grasslands is given maximum priority. There would be a push to extend the grasslands back to their pre-1905 extent. The long term viability of the grasslands will be increased under this scenario.

An economic approach to assess which of these three scenarios would be most desirable for the Culm Grasslands requires a quantification of the costs and benefits. In this section we present a simplified analysis, with full details being presented in Burgess et al. (2004) The cost benefit analysis is restricted to the economic benefits of increasing the landscape, ecology and recreational characteristics of the Culm Grasslands, relative to their continued decrease due to land use change and reducing ecological viability.

In order to assess the benefits of implementing an expansion scheme a contingent valuation (CV) study was undertaken. The study consisted of a survey which attempted to capture the values which members of the public might hold for the preservation of the Culm Grasslands, obtained through face-to-face interviews throughout Devon and Cornwall. Respondents were asked their willingness to pay, in the form of extra household taxation, for a 10% expansion of the Culm Grasslands through the restoration of converted and neglected land adjacent to the existing grasslands, in the expectation that this would provide protection and enhancement of the ecological, landscape and recreational characteristics of the area.

The final questionnaire was applied through interviews with residents and tourists at a range of locations throughout Devon and Cornwall. 247 surveys were completed. All interviewees were asked a one and a half bound elicitation question (Cooper et al. 2002). The one and a half bound elicitation method initially presents the respondent with a range of costs for implementing the scheme (BID_L for the lower cost and BID_U for the higher cost), with the bid level of BID_L and BID_U varied across the sample. Respondents were then asked whether they would be willing to pay either the lower cost or the higher one and depending on the response a second question would be asked. If the higher amount was offered first and the respondent indicates a positive WTP then no further WTP questions are asked, however if the respondent indicates they are unwilling to pay BID_U then the respondent would then be asked if they were willing to pay BID_L. Conversely, if respondents were initially offered BID_L a positive WTP would result in the respondent being asked whether they would pay BID_U, while a negative response would end the payment questions.

The results of the CV survey are shown in Tables D1 and D2. In order to obtain an estimate of the benefits derived from the 10% expansion of the Culm Grasslands, the individual WTP must be aggregated across the population. Aggregating these mean WTP values gives an annual benefit to the local and tourist populations of £12.6 million and £28.7 million respectively. Over the 10 year time horizon of the project the total value of expanding the Culm Grasslands by 10% will be £343.5 million[9].

[9] Applying a 3.5% discount rate, currently used by the government to evaluate the social projects and policies (HM Treasury, 1997)

Table D1: Mean and median individual WTP values based on a non-parametric model

	WTP (£ per household per year)	Confidence Interval (95%)
Median (WTP)	£15 to £25	-
Mean (WTP)	25.62	£19.16 – 32.08

Table D2: Mean WTP of the local and tourist sub-samples

	WTP (£ per household per year)	Confidence Interval (95%)
Mean (WTP) locals	24.51	£17.18 – 31.84
Mean (WTP) tourists	26.57	£15.44 – 38.92

The costs incurred in any expansion scheme fall into two categories: management costs and restoration costs. Culm Grassland requires specific management in terms of light grazing regimes and/or mowing or burning to prevent reversion of these areas to scrub and woodland. Management can be undertaken through either agri-environmental schemes or land purchase and subsequent maintenance or a combination of both. The total net benefits of each scenario can be calculated by subtracting the management and restoration costs from the value of total benefits.

The results of the CSERGE cost benefit analysis is summarised in table D3 below. The present value of management and restoration costs are over a 10 year time horizon discounted at 3.5%. The costs vary depending due to differing land management options (for further details see Burgess et al., 2004). It can be seen that the Policy Target Scenario generates large net benefits and on the basis of economic efficiency a scheme to increase the area of the Culm Grasslands by 10% should be implemented.

Table D3: Net benefits (£2003 million) of implementing the alternative expansion schemes under the Policy Targets and Deep Green Scenarios.

	PT Scenario	DG Scenario
Benefits (B)	344	344
Costs (C)	5 - 35	101 - 444
Net benefits (B - C)	**309 - 339**	**-100 - 243**

Annex H: Wetland Functions: Overview of Empirical Studies

The overview of studies presented in the following table, classifies the studies in terms of the function-use provided by the wetland wherever possible. Due to the multipurpose nature of many studies, in many cases it is not possible to specify this precisely, if at all. Hence many of the studies are listed in terms of the more general socio-economic uses and benefits that they consider.

The overview uses the following categories of valuation techniques:

Valuation techniques
OM= Optimisation Models (Residual imputation or variant)
DF= Damage cost Approach
MV= Market based
RC= Replacement Cost Method
TC = Travel Cost
CV= Contingent Valuation
HP=Hedonic Pricing

Bibliographic study characteristcs	Study Characteristics										
Author(s)	Title	Bibliographical details	Year	Issue addressed in study/General Function-Use Identification	Valuation technique	Year of data collection	Measurement unit	Estimated value characteristics: Mean/Total	Water system: Groundwater/ surface water	Spatial scale	Country
Bateman, I., *et al.*	"A Contingent Valuation Study of the Norfolk Broads"	Report to the National Rivers Authority	1992	Average WTP to preserve present landscape. Function-Use: Habitat, Non-use value	CV		English pounds per person per year	Use values: 78-105 Non-use values of local population: 14.7 Non-use values of the rest of GB: 4.8.	Broads	Regional	United Kingdom
Bateman, I.J., I.H. Langford, R.K. Turner, K.G. Willis, and G.D. Garrod.	"Elicitation and Truncation Effects in Contingent Valuation Studies	*Ecological Economics,* 2, 161-179*.*	1995	Analysis of methods of eliciting WTP in a CV study of flood protection of a UK wetland.	CV	1991 August, September	English pounds/years		Wetland	Local	United Kingdom
Bergstrom, J.C., J.R. Stoll, J.P. Titre, and V.L. Wright	"Economic Value of Wetlands-Based Recreation"	*Ecological Economics,* 2, 129-147.	1990	Wetlands loss and recreational value Function-Use: Recreation	CV	1986-1987	Dollars, per user.	360	Wetlands	Regional	USA
Breaux, A.S. Faber and J. Day.	"Using Natural Coastal Wetlands Systems for Wastewater Treatment: An Economic Benefit Analysis".	*Journal of Environmental Management,* 44, 285-291	1995	Wetland value for waste treatment use. Function-Use: Industrial Supply	RC		Dollars per year per firm.	(a. Value represents annualised cost saving to the firm from using a more extensive discharge dispersion system on a 6.2 acre wetlands site: 26700; (b. Estimate is wetland's treatment value per acre, including all plants' capitalised cost savings and based on treatment systems with a 25 year lifetime (low estimate): 6231.	Wetlands	Local	USA

Broadhead, C., J.P. Amigues, B. Desaigues, and J. Keith	"Riparian Zone Protection: The Use of the Willingness to Accept Format (WTA) in a Contingent Valuation Study.	Paper presented at the World Congress of Environmental and Resource Economists in Venice, Italy.	1998	In 1997, a study was financed by the French Ministry of Environment to evaluate the costs of preserving riparian habitat on the banks of the Garonne River. The CVM was used to study households that currently own land on the banks of the river. More precisely, a WTA was used to estimate the loss to owners for no longer being able to farm riverbank areas activity. Results of this study are reported and analysed in this paper.	CV	1997	FF/ha/year	Mean WTA fro program 1373FF/ha.	River	Regional	France
Brouwer, R. and L.H.G. Slangen.	"Contingent Valuation of the Public Benefits of Agricultural Wildlife Management: The Case of Dutch Peat Meadow Land".	*European Review of Agricultural Economics,* 25, 53-72.	1998	To provide a conservative estimate of the public benefits of agricultural wildlife management on Dutch peat meadow land and to provide a monetary estimate of the public benefits of management agreements. Function-Use: Habitat, Rare or Endangered Species	CV	1994	Dutch guilders and years	WTP: South Holland/ Friesland/Limburg/total: 131.4/113.6/64.5/124.5	Ditch	Regional	Netherlands
Cooper, J. and J.B. Loomis	"Testing whether Waterfowl Hunting Benefits Increase with Greater Water Deliveries to Wetlands"	*Environmental and Resource Economics,* 3(6), 545-561	1993	Impact on recreational waterfowl hunting benefits of an increase in refuge water supplies to levels necessary for biologically optimal refuge management	TC	1990	US $ per acre-foot of additional water supply	0.93 -- 20.40 (OLS), 0.64 -- 14.05 (Poisson)	Wetlands	Regional	USA
Cooper, J.C.	"Using the Travel Cost Method to Link Waterfowl Hunting to Agricultural Activities".	*Cahiers d'Economie et Sociologie Rurales,* 35, 5-26	1995	Impact of contaminated irrigation run-off on waterfowl hunting benefits. Function-Use: Recreation, Agricultural Supply.	TC	1988	US $ per hunter day and total for Kesterson	55.41	Wetlands	Regional	USA

Cordell, H.K. and J.C. Bergstrom.	"Comparison of Recreation Use Values Among Alternative Reservoir Water Level Management Scenarios".	*Water Resources Research*, 29 (2), 249-258	1993	Recreational benefits of three water level management alternatives in comparison to other use values (hydropower, flood control, etc.) Function-Use: Flooding, Recreation, Hydro power Generation.	CV	1988/1989	US $, per individual (>=12 years old) for access to TVA reservoirs per year	41.70 -- 75.05	Lake (reservoir)	Regional	USA
Costanzo, R., S.C. Farber and J. Maxwell.	"Valuation and Management of Wetland Ecosystems",	*Ecological Economics,* 1, 335-361.	1989	Coastal wetlands in Louisiana. Function-Use: Commercial Fishing	MV	1983	Dollars per acre. Dollars per acre per year.	(a Present value of the marginal product of an acre of wetland through production of five commercial fishery products (brown and white shrimp, menhaden, oyster, and blue crab) is reported. 3% was used for discounting: 845; (b Estimated value of annual average product of an acre of marsh and open water area is reported. This estimate may overvalue the wetland since average product is generally lower than marginal product, the more appropriate.	Wetlands	Regional	USA
Dalecki, MG., J.C. Whitehead and G.C. Blomquist.	"Sample Non-Response Bias and Aggregate Benefits in Contingent Valuation: an Examination of Early, Late, and Non-respondents",	*Journal of Environmental Management*, 38: 133-143.	1993	Wetland preservation. Function-Use: Wetland Habitat	CV	1990	$/person/year	(a. Individual median WTP estimate for wetland preservation of the first wave (response rate = 24%0: 24.4; (b. Individual median WTP estimate for wetland preservation of the fourth wave (response rage = 67%): 6.54.	Wetlands	Regional	USA

Foster, V., I.J. Bateman and D. Harley.	"Real and Hypothetical Willingness to Pay for Envirnmental Preservation: A Non-Experimental comparison".	In Environmental Valuation, Economic Policy and Sustainability: Recent Advances in Environmental Economics. Melinda Acutt and Pamela Mason (eds.). Northampton, MA: Edward Elgar, 35-49.	1998	Land purchases, species preservaton, and habitat conservation. Function-Use: Habitat, Rare or Endangered Species.	MV	1995	Pounds sterling per mailing. Pounds Sterling	)a. Reported value is the mean donation per mailing to the RSPB fund raiser. The fund raising appeal was for the land purchase of maritime health habitat in Ramsey Island in 1992. This is the average donation (includes returned and not returned): £1.73/mailing; (b. Reported value is the total value of donations for the RSPB fund raiser. The fund raising appeal was for the portection of reedbed habitat for bittern in 1993: £268430.	Wetlands	National	United Kingdom
Gren, I.M.	"Alternative Nitrogen Reduction Policies in the Malar Region, Sweden",	*Ecological Economics,* 7(2), 159-172.	1993	Denitrification functions of wetlands. Function-Use: Habitat	RC	1991	SEK millions (1US $ = SEK 5.8). SEK/Kg N.	(a. Value is the total cost of restoring wetlands that reduce the load of nitrogen by 1,194 tons. Significant cost reduction for nitrogen abatement can be attained through restoring wetlands: 49; (b. Value is the high-end estimate for the marginal cost of abating 1 Kg of nitrogen through restoring wetlands. Significant cost reduction for nitrogen abatement can be attained through restoring wetlands.	Wetland	Regional	Sweden
Gupta, T.R. and J.H. Foster	"Economic Criteria for Freshwater Wetland Policy in Massachusetts",	*American Journal of Agricultural Economics*, 57(1), 40-45.	1975	Function-Use: Recreation. Multiple uses/benefits associated with wetlands (value of wildlife, visual-cultural benefits, water supply, and flood control benefits of wetlands).	DF	1972	Dollars per acre per year.	(a. Value represents average benefits from flood control for low quality acres: 10; (b. Value represents average benefits from flood control for high quality acres: 80>	Wetlands	Regional	Jordan

Heimlich, R.E.	"Costs of an Agricultural Wetland Reserve",	*Land Economics,* 70(2), 234-46.	1994	Wetlands converted from cropland.	RC	1982	Dollars per acre.	(a. Value is the high estimate of the marginal costs of a 5 million acres of wetland reserve: 1184; (b. Value is the high estimate of the total average cost (in $/acre) that minimizes reserve costs for wetland reserve of 1 million acres: 286.	Wetlands	National	USA
Klein, R.J.T. and I.J. Bateman	"The Recreation Value of Cley Marshes Nature Reserve: An Argument against Managed Retreat?	*Water and Environmental Management,* 12, 280-285.	1998	The main aim of this study is to provide an estimate of the recreational value of the Cley Reserve. Function-Use: Recreation, Habitat	CV	1996	A: In UK pounds, per household, per year or per visit. B. In UK pounds, per party per annum	WTPfee (incl. Zero-bids, in UK pounds): 1.58; WTPfee (excl.): 2.22; WTPtax (incl.): 48.15; WTPtax (excl.): 62.08.	Reserve	Regional	United Kingdom

Kosz, M.	"Valuing Riverside Wetlands: The Case of the Conau-Auen" National Park".	*Ecological Economics,* 16, 109-127.	1996	The aim of this paper is briefly to review the main results of the cost-benefit analysis concerning all the variables that depend on direct anthropocentric use, including energy production with hydroelectric power stations, shipping, ground water protection, stabilisation of the river bed to stop channel erosion, visitors' benefits, forestry, farming, fishing, hunting and the costs of establishing a national park. This was done because there was a plan to build one or more hydroelectric power stations in the area under study, the Donau-Auen. This was operationalised by 4 different development projects (1) Establishing a national park in all easily available areas (not included in the WTP value. (2) Founding a national park in all available areas including private property; concept of hydraulic engineering including extensive measures artificially changing the waterway to avoid further river bed erosion. (3) Construction of a hydroelectric power station near Wolfsthal. (4) Construction of a hydroelectric power station near Wildungsmauer. (The last project is higher in magnitude compared to the third).	CV	1993 (June and July)	ATS 1993 a year	2a) 919, 80; 2b) 329, 25; 3a) 694,9; 3b) 122,21: 4a) 689,85; 4b) 69, 63	River	Regional	Austria
Mannesto, G. and J.B. Loomis	"Evaluation of Mail and In-Person Contingent Value Surveys: Results of a Study of Recreational Boaters",	*Journal of Environmental Management,* 32, 177-190.	1991	Wetland loss Function-Use: Recreation	CV	Interview data drom 29 August to 9 October 1987; Mailing data also in this same period.	$ and concerning the mail back list: 25% increase or 50% increase of total delta wetlands.	1a) 69.80; 1b) 37.12; 1c) 37.85; 2a) 59.27; 2b) 39.47; 2c) 33.14.	Delta LAKE BAY	Regional	USA

Miyata, Y. and H. Abe.	"Measuring the Effects of Flood Control Project: Hedonic Land Price Approach.	*Journal of Environmental Management,* 42, 389-401.	1994	Aim is to measure the effects of a flood control project planned for the Chitose River Basin in Japan by evaluating the reduction in expected physical flood damage derived by construction and improvement of flood control facilities. Function-Use: Flooding.	HP	1990	Yen per Km^2 cm and unit area.	The total annual average cost of the flood control project for the Chitose River (in million yen): case 1: project cost/annual average cost: 0/0; case 2: 96787/4898; case 4: 201848/10214; case 5: 267405/13531; case 6: 310366/15705. Tot al benefit: Ebvetsu; 5032.0/146.3;Chitose; 12499.2/336.0;Eniwa: 24460.3/497.2; Hiroshima: 8191.5/615.9;Nanporo: 7479.2/138.2; Naganuma: 26390.2/288.4; total: 84052.4/300.5. The corresponding total cost is estimated as 310.4 billion yen and the total estimated benefit computed from the land price variations is 84 billion yen, thus the flood control project under this study may be deemed as a less cost-efficient project.	River basin. Catchment	Regional	Japan

Roberts. L.A. and J.A. Leitch.	"Economic Valuation of Some Wetland Outputs of Mud Lade, Minnesota, South Dakota".	Agricultural Economics Report No. 381, Department of Agricultural Economics, North Dakota State University, USA	1997	The purpose of this study was to approximate some economic values of Mud Lake, a managed "wetland" on the border between Minnesota and South Dakota, to provide information to promote more efficient and effective management of Mud Lake and its wetlands. This is done by evaluating some selected outputs: flood control, water supply, fish and wildlife habitat, recreation and aesthetics, and disamenities to water quality. The DVM was used to valuate fish and wildlife habitat, recreation, and aesthetics. Water quality was valued by estimating the extra costs of water treatment, flood control by damages prevented, and water supply by estimating a residual return to public wear utilities. Function-Use: Recreation, Flooding.	CV	1995	$ per year per acre	Flood control: total: $440; Water supply/conservation: $94; WTP regarding fish/wildlife habitat, recreation, and aesthetics: 1) $7;2) $8, 3) $6.	Lake	Regional	USA
Stevens, T.H., S. Benin and J.S. Larson	"Public Attitudes and Economic Values for Wetland Preservation in New England.	*Wetlands*, 15(3), 226-231.	1995	Wetlands in New England. Function-Use: Flooding	CV	1993	Dollars per respondent.	(a. Value is the high end estimate of respondents' yearly WTP to protect New England wetlands that provide flood protection, water supply and pollution contorl: 80.41; (b. Value is the low end estimate of respondents' yearly WTP to protect New England wetlands that provide flood protection, water supply and pollution contorl: 73.89.	Wetlands	National	New England

Steever, W.J., M. Callaghan-Perry, A. Searles, T. Stevens and P. Svoboda.	"Public Attitudes and Values for Wetland Conservation in New South Wales, Australia".	*Journal of Environmental Management,* 54(1), 1-14.	1998	Wetland conservation. Function-Use: Habitat	CV	1996	Australian dollars/person/ year for 5 years.	(a. Value represents median WTP for the pooled sample. Value from the pooled sample omits those rspondents who did not express WTP: 100; (b. Value represents aggregate value for wetlands in New South Wales, Australia, assuming a WTP per household of A$17.10 and 2.23 million jhouseholds in the state: 38.	Wetlands	Regional	Australia
van Kooten, G.C.	"Bioeconomic Evaluation of Government Agricultural Programmes on Wetland Conversion",	*Land Economics,* 9(1), 27-38.	1993	Wetlands providing migratory waterfowl habitat and recreation opportunities. Function-Use: Agricultural Supply.	OM	1988	Dollars per acre per year.	Marginal value of waterfowl habitat as cropland per acre year is reported. Government subsidy of $4.50 per bushel of grain and an average yield of 30 bushels/acre were assumed (land has no livestock value): 37.97.	Wetland	Regional	USA
Whitehead, J.C.	Measuring Willingness to Pay for Wetlands Preservation with the Contingent Valuation Method.	*Wetlands,* 10(2), 187-201.	1990	Preservation of a bottomland hardwood forest wetland. Function-Use: Habitat	CV	1989	$/household/ year.	Value measures mean WTP for wetland preservation estimated from log-linear form of model: 6.31.	Wetland	Local	USA
Whitehead, J.C.	"Environment Interest Group Behaviour and Self-Selection Bias in Contingent Valuation Mail Sruveys"	*Growth and Change,* 22(1), 10-21.	1991	Wetland preservation. Function-Use: Habitat	CV	1989	$/person/year.	(a. Value is the average WTP per person/year in the general sample for the preservation of the Clear Creek wetland area (assuming 15% of the general population belongs to an environmental interest group): 4.12; (b. Value is the average WTP per person/year in the environmental interest group sample for the preservation of the Clear Creek wetland area: 42.83.	Wetland	Local	USA

Willis, K.G.	"Valuing non-market wildlife commodities: An evaluation and comparison of benefits and costs".	*Applied Economics,* 22, 13-30.	1990	WTP for the preservation of the current state of the wetlands. Function-Use: Recreation, Habitat	CV		£/ha.	(a. total use value: 44; (b total non-use value: 807.	Wetlands	Regional	United Kingdom

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