ENGLISH NATURE

Ecosystem functions and the implications for economic evaluation

No. 441 - English Nature Research Reports

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English Nature Research Reports

Number 441

Ecosystem functions and the implications for economic evaluation

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ISSN 0967-876X

Acknowledgements

We would like to thank our collaborators in the EU funded projects FAEWE and ECOWET:

Ros Boar, Tony Davy, Hans Schutten, Tom Crowards

Obviously, the authors of this report remain solely responsible for the content of this paper

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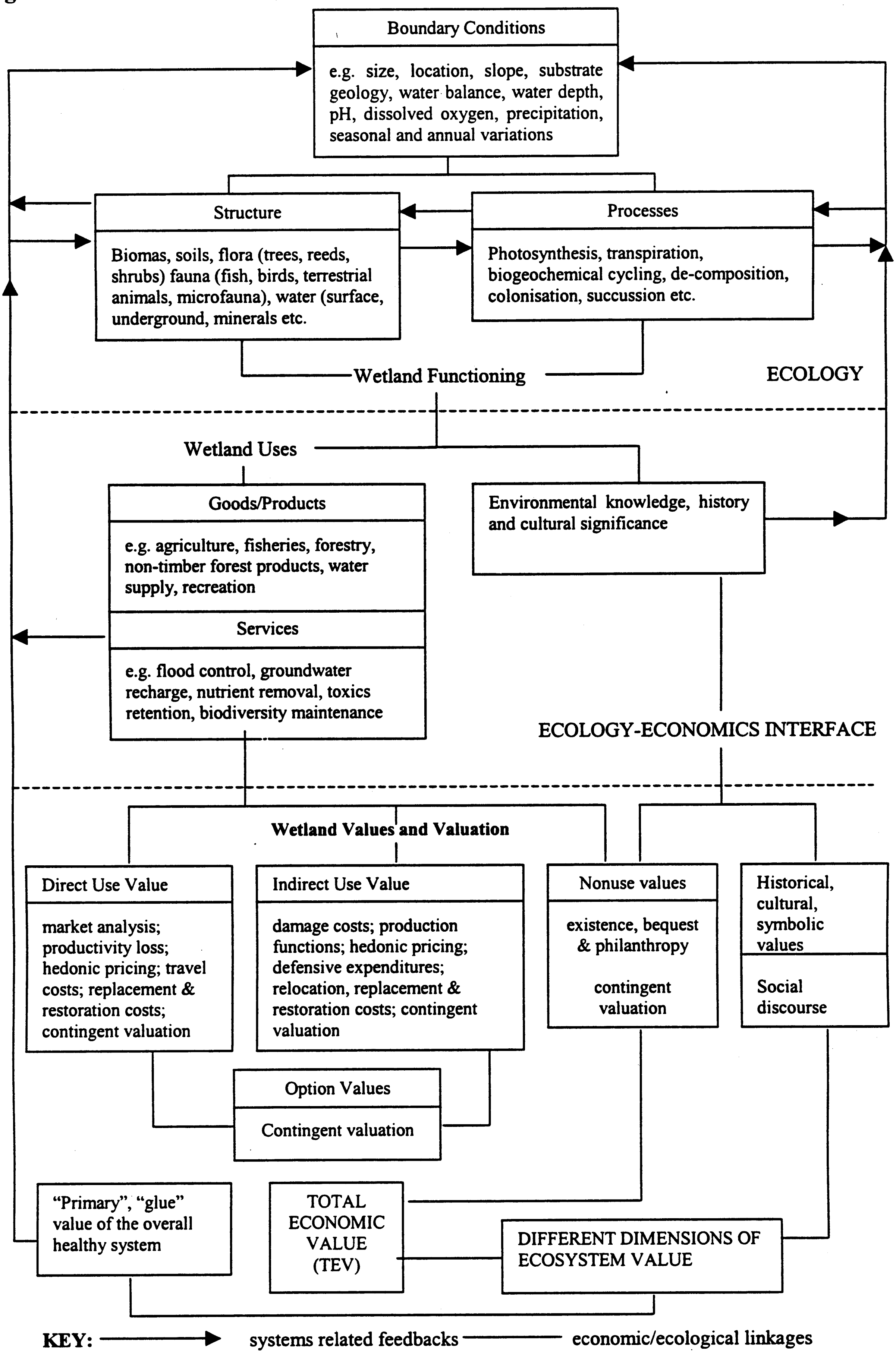
Introduction

Environmental systems (natural capital) have been experiencing intense and sustained environmental pressure and stresses from a range of direct and indirect socio-economic driving forces. Given this context, biodiversity conservation has to be interpreted as efforts to manage the rate of environmental change. Responsible agencies are therefore seeking better ways of managing the causes and consequences of the environmental change process across a range of sites and landscapes. Given the generic goal of sustainable development, management agencies should seek to maintain the resilience of systems, in terms of their ability to cope with stress and shock, and thereby enhance capacities that allow adaptation to both physical and social vulnerability. System resilience maintenance and/or enhancement is linked to the ecological concept of functional diversity and the social science analogue, functional value diversity. The latter concept combines ecosystem processes, composition and functions with outputs of goods and services, which can then be assigned monetary economic and/or other values (see Figure 1). A management strategy based on the sustainable utilisation of ecosystems should have at its core the objective of ecosystem integrity maintenance i.e. the maintenance of systems components, interactions among them ('functioning') and the resultant behaviour or dynamic of the system. The strategy must therefore adopt a relatively wide perspective and examine and seek to manage larger-scale (landscape) ecological processes, together with the relevant environmental and socio-economic driving forces. Such an approach is consistent with the UK's Biodiversity action plans, 'regional profiles' and English Nature's "natural areas" designation.

Most managed ecosystems are complex and often poorly understood hierarchically organised systems. Capturing the range of relevant impacts on natural and human systems under different management options will be a formidable challenge. Biodiversity has an hierarchical structure which ranges from the ecosystem and landscape level, through the community level and down to the population and genetic level. There is a need to develop methodologies for the practicable detection of ecosystem change, as well as the evaluation of different ecological functions. What is also required is a set of indicators (environmental, social and economic) which facilitate the detection of change in ecosystems suffering stress and shock and highlight possible drivers of the change process. A hierarchical classification of ecological indicators of sustainability would need to take into account existing interactions between different organisation levels, from species to ecosystems. Effects of environmental stress are expressed in different ways at different levels of biological organisation and effects at one level can be expected to impact other levels, often in unpredictable ways.

Figure 1: Wetland functions, uses and values



Another stage in the appraisal process involves the use of evaluation methods and techniques to assess on sustainability grounds whether any given management strategy is supporting, or reducing, the diversity of functions which are providing stakeholders with the welfare benefits they require.

Towards an environmental decision support system

Management actions need to be underpinned by a scientifically credible but also pragmatic environmental decision support system i.e. a toolbox of evaluation methods and techniques, complemented by a set of environmental change indicators and an enabling analytical framework. The support system should allow managers to identify a number of steps or 'decision rules' in order to operationalise the framework in a given natural habitats context, in this report wetland ecosystem management.

The following steps are recommended in the appraisal process:

- **Scoping and problem auditing**

The DP-S-I-R (driving pressures - state - impacts - response) framework, originally developed by the OECD, is a useful device for the scoping of biodiversity management issues and problems (see Figure 2). It can assist in determining the causes of habitat/species degradation or loss and the links to socio-economic activities, across the relevant spatial and temporal scales. It also provides the important conceptual connection between ecosystem change and the effects of that change (impacts) on people's economic and social wellbeing. Loss of ecosystem function provision is terms of goods and services (direct and indirectly received) can be translated into human welfare loss and quantified in monetary and/or other more qualitative ways (see Figure 1.) The DP-S-I-R framework is an auditing process, not a model. It main purpose is to make tractable the complexity of environmental problems and provide a basis for the derivation of relevant indicators of environmental change (see Figure 3).

- **Identification and selection of appropriate decision-making calculi-research methods**

Managed ecosystems will be in an almost constant state of flux as the natural processes and systems react to human management interventions, which in turn, subject to various lags, produce more policy responses, ie a coevolutionary process characterised by continuous feedback effects. It is therefore important to be able to assess the impact of alternative sets of management actions or strategies in order to judge their social acceptability against a range of criteria such as environmental effectiveness, economic efficiency, fairness across different stakeholder interests (including different generations) etc. Evaluation methods and techniques have to be matched up to the chosen evaluation criteria.

One key to valuing a change in an ecosystem function is establishing the link between that function and some good/service flow valued by people. The mainstream economic concept of total economic value (TEV) is composed of two dimensions of value, production and individual preference values. Production values of biodiversity are arguments in the production and cost functions of market allocated goods. These production inputs affect individual welfare via changing prices of goods or other inputs. Individual values on the other hand, are a direct argument of individual utility (well-being) functions. But these two dimensions of value can be supplemented with others: there is the overarching infrastructural role that biodiversity performs ("primary value" in nature) as it "glues" system components together; secondly, the socio-cultural and historical contexts in which environmental assets exist provide for alternative aspects of environmental value which may not be captured by the market paradigm; finally from the "deep ecology" worldview nature possesses "intrinsic value" which exists regardless of human use or appreciation (see Figure 4).

- **Data collection and monitoring via indicators**

The data required for monitoring environmental change can be conveniently classified in terms of three dimensions of value outlined in Figure 1 - primary/glue value possessed by ecosystems; total economic value assigned to ecosystem function outputs; and the socio-cultural, historical and symbolic value inherent in some environmental assets.

Primary value-related data and indicators

Primary value-related data collection should be based on an overall systems approach and ecosystem integrity in terms of structure, composition and functioning. Both quantitative and qualitative descriptive indicators of ecosystem integrity are required because of the level of uncertainty that surrounds the scientific understanding of how complex systems really work. This lack of knowledge also means that a precautionary approach to ecosystem conservation is recommended, with normative benchmarks to assess the sustainability of systems and management regimes.

Ecosystem integrity can be defined as the maintenance of system components, interactions among them and the resultant behaviour or dynamic of the system. The 'integrity' of an ecosystem is more than its capacity to maintain autonomous functioning (its health); it also relates to the retention of 'total diversity' i.e. the species and interrelationships that have survived over time at the landscape level. The focus here therefore, is on the importance of the overall system and its 'infrastructural' value in providing enough of an operational system (e.g. a wetland) to guarantee the provision of a range of functions and related goods and services. The resilience capacity is also related more to overall system configuration and stability properties, than it is to the stability of individual components (albeit with due recognition to particular keystone species and/or processes). This has important implications for the assessment of individual conservation sites which within their own boundaries may service only a single function or host a single 'important' species. The 'value' of such sites can only be properly addressed when the site is viewed within the larger landscape ecosystem and its contributory value is recognised. The 'natural areas' approach and the 'regional profiles' within the Biodiversity Action Plan championed by English Nature fit the overall systems perspective which is recommended here.

Figure 2: DP-S-I-R Framework: Continuous Feedback Process

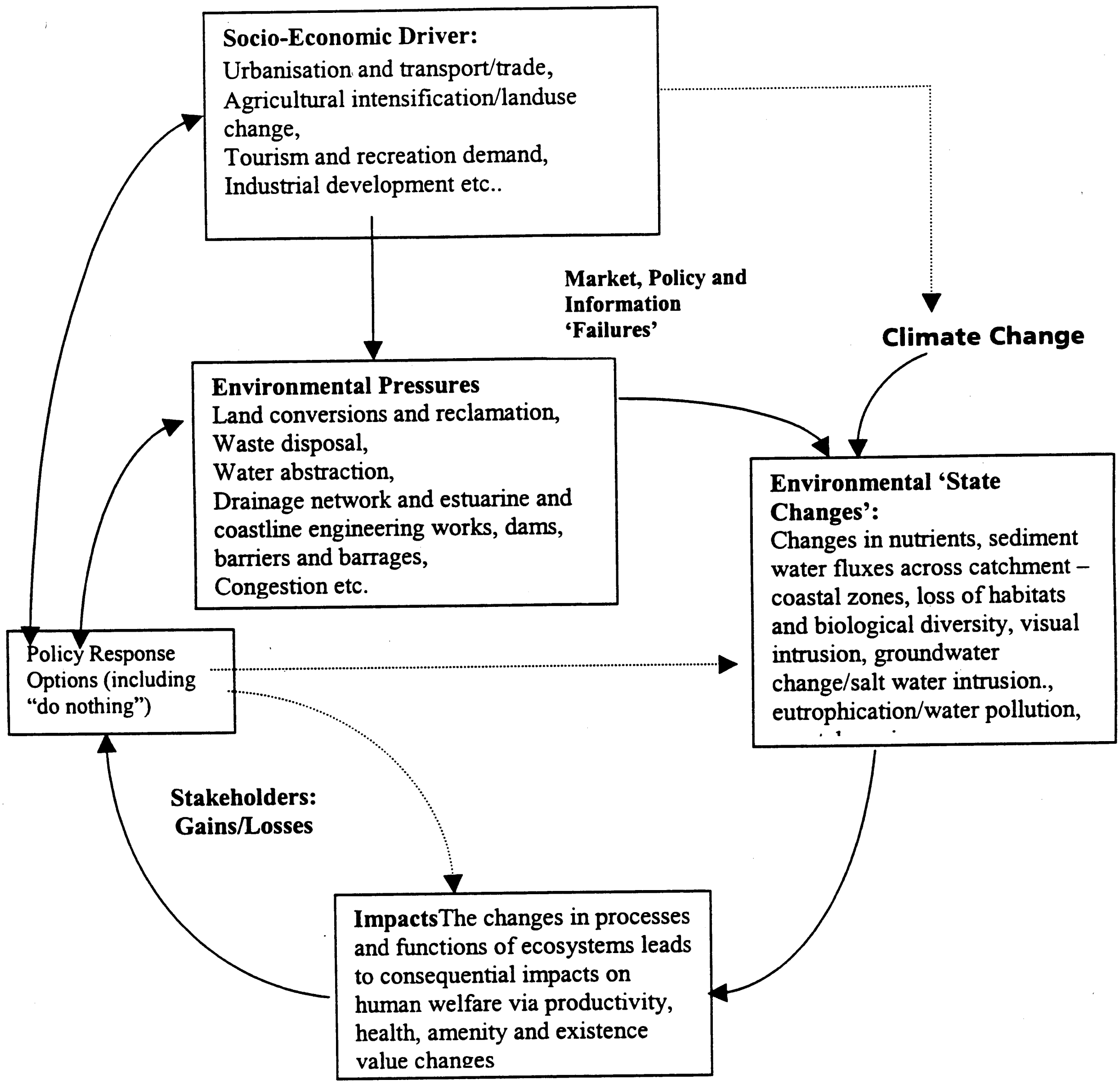
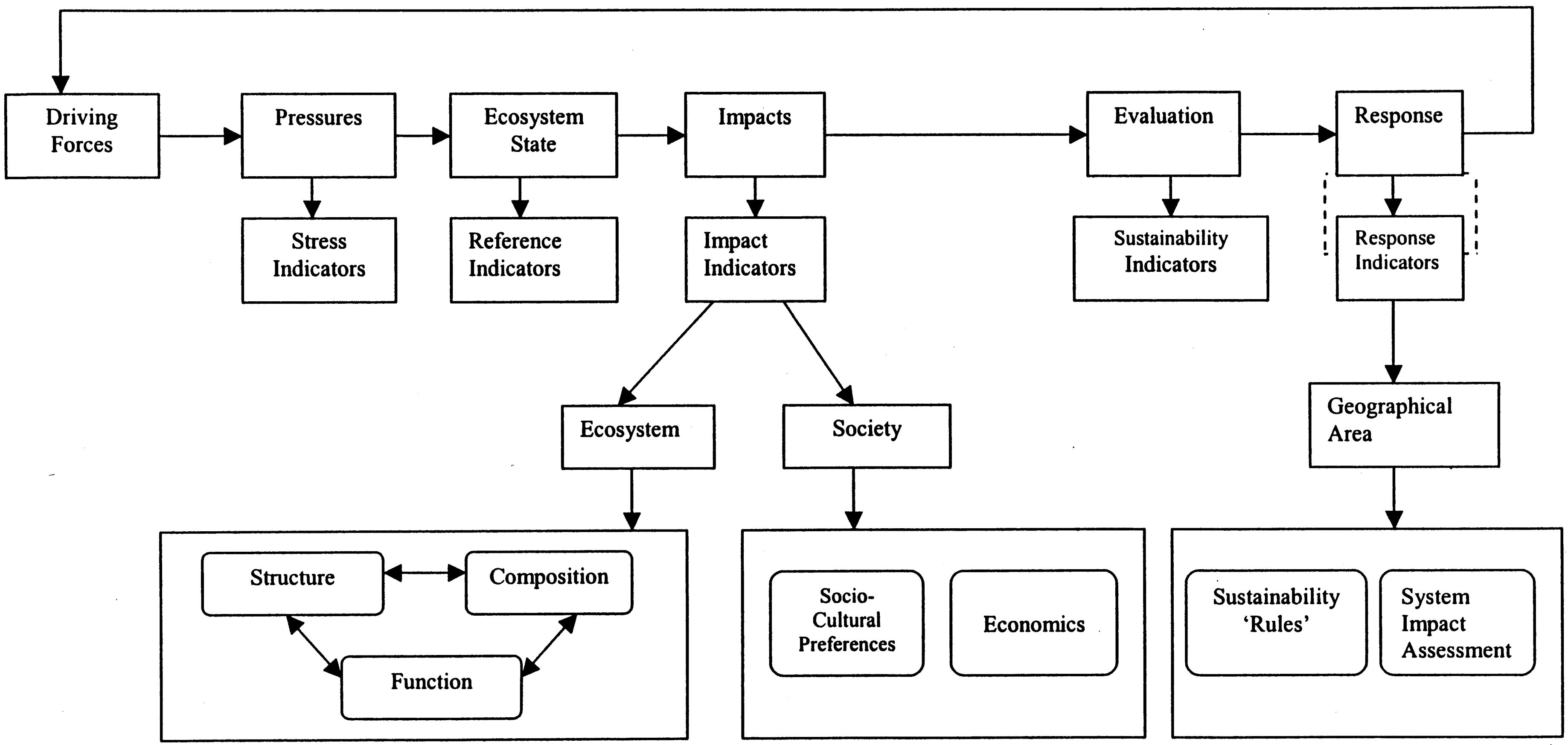


Figure 3. The derivation of environmental indicators to evaluate the sustainability of natural resource management: an extended DPSIR model



The effect of pressures exerted by human activities on ecosystems can be measured by defining the relevant indicator spheres for ecosystem structure, composition and functioning. This breakdown, originally proposed in the context of biodiversity assessment techniques, can be used to organise the indicator sets which cover three interrelated aspects of ecosystems: landscape, water regime and biodiversity (see figure 5). This holistic approach focuses on the interdependency and compatibility within and between indicator sets across different scales.

Total economic value-related data

Both the socio-economic and natural scientific aspects of ecosystem integrity are integral to our approach. The environmental indicators must be analysed and evaluated vis-á-vis the social context in which they arise. This context includes the institutional, political, socio-cultural and spatial/temporal scale, as well as the economic circumstances through which environmental change occurs and is monitored.

In order to consider ecosystem components valuation we rely on the concept of functional value diversity. This links ecosystem processes and functioning with outputs of goods and services which can be assigned monetary economic values. From an anthropocentric viewpoint all ecosystems can be classified in terms of their structural and functional aspects. Ecosystem structure is defined as the tangible items such as plants, animals, soil, air and water of which it is composed. Thus structural benefits (of instrumental value to humans) include fish, waterfowl, peat, timber, reed and fur harvests as well as non-consumptive use benefits such as recreation and research or education. By contrast, ecosystem processes are encompassed by the dynamics of exchange or means of energy. The processes are subsequently responsible for the services - life support services, such as assimilation of pollutants, cycling of nutrients and maintenance of the balance of gases in the air (see Table 1 for an application to wetlands).

The diversity concept also encourages analysts to take a wider 'landscape ecology' perspective in terms of examining changes in large scale (land use level) ecological processes, together with the relevant socio-economic driving forces causing ecosystem loss. The focus is then on the ability of interdependent ecological-economic systems to maintain functionality under a range of stress and shock conditions i.e. it relates back to the concepts of ecosystem integrity and resilience. Functional diversity can be defined as a variety of different responses to environmental change, in particular the variety of spatial and temporal scales with which organisms react to each other and to the environment. The policy objective of maximum diversity maintenance serves to ensure the maximum amount of functional value in terms of goods and services provision. The model requires the practical coupling of economic, hydrological and ecological models.

There are a number of key issues and ecological principles relating to the functioning of ecosystems and the assignment of values to ecosystem structure and functions.

1. *Spatial and Temporal Scale* - Ecological processes operate over a range of spatial and temporal scales; scales of time or space appropriate for the study of management of one process may not be suitable for other processes.

2. *Ecosystem Function Depends on Structure, Complexity and Diversity* - Biological diversity underlies the complexity of ecosystem function.

Ecosystem productivity is hypothesised to be higher when more plant species are present because differences among species in methods of resource capture should allow more diverse communities to more fully utilise their limiting resources.

Biological diversity also provides for both stability (resistance) and recovery from disturbances that disrupt important ecosystem processes.

Diversity, furthermore, provides long term capacity for adaptation, as well as being a sensitive indicator of environmental change. It is important to acknowledge that the contribution of diversity to ecosystem functions is not merely a function of the number of species. Their identity is also important.

3. *Ecosystems are Dynamic in Space and Time* - Change is the normal course of events for ecosystems. Natural or human induced disturbance on landscapes creates a patchwork mosaic, and the resulting changes initiated within each patch are influenced by the pattern and behaviour of surrounding patches. This landscape variation influences ecosystem functioning at large spatial scales.

4. *Uncertainty and Surprise are Inevitable* - There is much that is not understood about ecosystems. Some of that ignorance will yield to increased knowledge, but the complexity and interactions of non-linear processes means that certain elements of ecosystem function will always be difficult to predict and that surprises in ecosystem behaviour are inevitable. This is particularly the case with the managed usage ecosystems.

The essence of an overall socio-economic evaluation is to determine how society is affected by the functions an ecosystem might perform - the function itself is not instrumentally valuable. Functions in themselves are therefore not necessarily of economic value; such value derives from the existence of a demand for these functions or for the goods and services they provide. It is therefore important to identify how particular functions might be of use, rather than simply the degree to which the function is being performed. The extent of demand for the products or services provided, or the effective 'market', needs to be assessed if the full extent of economic value is to be assessed. It will therefore be necessary to assess features of socio-economic activities and behaviour and how these respond to changes in ecosystem functioning.

The key to valuing a change in an ecosystem function is establishing the link between that function and some service flow valued by people. If that link can be established, then the concept of derived demand can be applied. The value of a change in an ecosystem function can be derived from the change in the value of the ecosystem service flow it supports. However, the multifunctional characteristic of ecosystems makes comprehensive estimation of every function and linkages between them difficult.

A number of ecosystem goods and services can be valued in economic terms, while others cannot because of uncertainty and complexity conditions. Ecosystems provide a wide range of goods and services of significant value to society - storm and pollution buffering function, flood alleviation, recreation and aesthetic services, etc. We can therefore conceive of 'valuing' an ecosystem as essentially valuing the characteristics of a system, and we can capture these values in our economic value framework. But since it is the case that the component parts of a system are contingent on the existence and continued proper functioning of the whole, then putting an aggregate value on ecosystems is quite a complicated matter.

In instrumentally valuing a resource such as an ecosystem, the total economic value (TEV) can be usefully broken down into a number of categories. Use value involves some interaction with the resource, either directly or indirectly:

1. *Indirect use value* derives from services provided by the ecosystem. This might for example include the removal of nutrients, providing cleaner water to those downstream, or the prevention of downstream flooding.

2. *Direct use value*, on the other hand, involves interaction with the ecosystem itself rather than via the services it provides. It may be consumptive use such as the harvesting of reeds or fish, or it may be non-consumptive such as with some recreational and educational activities. There is also the possibility of deriving value from □distant use□ through media such as television or magazines, although whether or not this type of value is actually a use value, and to what extent it can be attributed to the ecosystem involved, is unclear.

3. *Non-use value* is associated with benefits derived simply from the knowledge that a resource, such as an individual species or an entire ecosystem, is maintained. It is by definition not associated with any use of the resource or tangible benefit derived from it, although users of a resource might also attribute non-use value to it. Non-use value is closely linked to ethical concerns, often being linked to altruistic preferences. It can be split into three basic components, although these may overlap depending upon exact definitions:

- ***Existence value* can be derived simply from the satisfaction of knowing that some feature of the environment continues to exist, whether or not this might also benefit others. This value notion has been interpreted in a number of ways and seems to straddle the instrumental/intrinsic value divide.**
- ***Bequest value* is associated with the knowledge that a resource will be passed on to descendants to maintain the opportunity for them to enjoy it in the future.**
- ***Philanthropic value* is associated with the satisfaction from ensuring resources are available to contemporaries of the current generation.**

4. Option value, in which an individual derives benefit from ensuring that a resource will be available for use in the future. In this sense it is a form of use value, although it can be regarded as a type of insurance to provide for possible future but not current use.

5 Quasi-option value is associated with the potential benefits of awaiting improved information before giving up the option to preserve a resource for future use. It suggests a value in particular of avoiding irreversible damage that might prove to have been unwarranted in the light of further information. An example of an option value is in bio-prospecting, where biodiversity may be maintained on the off-chance that it might in the future be the source of important new medicinal drugs.

Valuation techniques

A range of valuation techniques exists for assessing the economic value of the functions performed by ecosystems (see Table 2). Many ecological 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 non-market valuation techniques.

Why the value of ecosystems can be greater than total economic value

The social value of an ecosystem, may not be equivalent to the aggregate private total economic value of that same system's components, for the reasons described below:

Complexity: The full complexity and coverage of the underpinning 'life-support' functions of healthy evolving ecosystems is currently not precisely known in scientific terms. A number of indirect use values within systems therefore remain to be discovered and valued (quasi option value i.e. the conditional expected value of information).The task of evaluating the structure and function of an ecosystem implies that we know fully what the ecosystem does and what that worth is to us. The worth of ecosystem structure is generally more easily appreciated than that of ecosystem functions. To evaluate functions such as nitrogen fixation, nutrient and soil retention, gas exchange, radiation balance, pollution absorption, and other ecological processes for any given segment of landscape, pushes present ecological knowledge beyond its bounds. Even ecosystem structure is incompletely known. To evaluate the worth of the insect fauna, or soil fungi, when many of these species have never even been described taxonomically, taxes human knowledge beyond current limits.

Redundancy reserve: A healthy ecosystem also contains a redundancy reserve, a pool of latent keystone species/processes which are required for system maintenance in the face of stress and shock. This is what the quasi-option value concept seeks to capture and in this report it is interpreted to mean more than an ex post option value judgement.

Primary and glue value: Because the range of secondary values (use and non-use) that can be instrumentally derived from an ecosystem is contingent on the prior existence of such a healthy and evolving system, there is in a philosophical sense a 'prior value' that could be ascribed to the system itself. Such a value would, however, not be measurable in conventional economic terms and is non-commensurate with the economic (secondary) values of the system. The continued functioning of a healthy ecosystem is more than the sum of its individual components. There is a sense in which the operating system yields or possesses 'glue' value, i.e. value related to the structure and functioning properties of the system which hold everything together.

Figure 4. Approaches to Valuation

Simplified Typology

[Mainstream Position]	[Extended CBA Position]	[Environmental Management without prices]
Individualism	environmental systems-functions	preferences not appropriate basis for valuation
Exogenous preferences	individualism vs collectivism-consumer and	rights-based approaches environmental 'trump'
fixed preferences	endogenous preferences-psychosocial and	citizen motivations as a distinct and separate
marginal/discrete environmental changes	expressed and revealed preferences	keep 'markets' out of the environment
revealed preferences via markets; household production functions	focus group testing	expert opinion
positive rates of discount	benefits transfer	contingent valuation as opinion polls
efficiency criterion	non-use values	deliberative and inclusionary processes to assign
economic welfare	validity/reliability testing protocols	
	equity criterion-policy trade-offs econ. welfare	
	standards/regulations-cost effectiveness	

Table 1: Wetland functions and associated socio-economic benefits

Function	Biophysical Structure or Process Maintaining Function	Socio-Economic Use and Benefits	Threats
Hydrological Functions			
flood water retention	short and long term storage of overbank flood water and detention of surface water runoff from surrounding slopes	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
groundwater recharge	infiltration of flood water in wetland surface followed by percolation to aquifer	water supply, habitat maintenance	reduction of recharge rates, overpumping, pollution
ground water discharge	upward seepage of ground water through wetland surface	effluent dilution	drainage, filling
sediment retention and deposition	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	improved water quality downstream, soil fertility	channelization, excess reduction of sediment throughput
Biogeochemical Functions			
nutrient retention	uptake of nutrients by plants (N and P), storage in soil organic matter, absorption of N as ammonium, absorption of P in soil	improved water quality	drainage, water abstraction, removal of vegetation, pollution, dredging
nutrient export	flushing through water system and gaseous export of N	improved water quality, waste disposal	drainage, water abstraction, removal of vegetation, pollution, flow barriers
peat accumulation	in situ retention of C	fuel, Paleo-environmental data source	overexploitation, drainage
Ecological Functions			
habitat for (migratory) species (biodiversity)	provision of microsites for macro-invertebrates, fish, reptiles, birds, mammals and landscape structural diversity	fishing, wildfowl hunting, recreational amenities, tourism	overexploitation, overcrowding and congestion, wildlife disturbance, pollution, interruption of migration routes, management neglect
nursery for plants, animals, micro-organisms	provision of microsites for macro-invertebrates, fish, reptiles, birds, mammals	fishing, reed harvest	overexploitation, overcrowding and wildlife disturbance, management neglect
food web support	biomass production, biomass import and export via physical and biological processes	farming	conversion, extensive use of inputs (pollution)

Figure 5: Ecosystem integrity indicator spheres

Ecosystem feature	Ecosystem attribute	Indicator sphere
	Structure	wetland area structure
Landscape	Composition	land form and cover
	Function	land use changes
	Structure	hydrology
Water regime	Composition	biogeochemical water properties
	Function	water constituents flux
	Structure	food web trophic structure
Biodiversity	Composition	keystone species and umbrella species
	Function	energy transfer between trophic levels

The recognition of complementary relationships implies that the total value of biodiversity is infinite. As a basis of human life, it is indispensable under realistic technological and economic conditions. Marginal decisions or apparently marginal decisions as perceived by different stakeholders, are the stuff of the real world political economy, and therefore need to be considered. The problem with them is that knowledge about the consequences of infringements on biodiversity is incomplete. There is an unbridgeable gap in our knowledge about ecosystem interrelationships and regularities. The benefits of biodiversity protection will often only be discovered, once they have been disturbed or lost.

Socio-cultural, symbolic value data

The task of sustainable management can be defined as sustainable utilisation of the multiple goods and services generated by ecosystems, together with the 'socially equitable' distribution of welfare gains and losses inherent in such usage. However, the social welfare account will include not just economic welfare stocks and flows but also changes in properties such as sense of identity, culture and historical significance in ecosystem components and overall landscapes. Compiling values data in this context is likely to be more 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 then assumed to give rise to different degrees of support for alternative decision-making procedures and the underlying valuations elicited via the social discourse process.

Table 2. Valuation Methodologies Relating to Ecosystem 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 (does-response) market analysis.	√	√	
(Production Functions)	Wetland treated as one input into the production of other goods: based on ecological linkages and market analysis.		√	
(Public Pricing)	Public investment, for instance via land purchase or monetary incentives, as a surrogate for market transactions.	√	√	√
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)	Cost 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 wetland function were not present; eg 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'.	√	√	√
Restoration Costs	Costs of returning the degraded wetland to its original state. A total value approach; important ecological, temporal and cultural dimensions.	√	√	√

Notes:

[1] Indirect use values associated with functions performed by a wetland 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 wetland.

[2] Investment by public bodies in conserving wetlands (most often for maintaining biodiversity) can be interpreted as the to the total value attributed to the wetland by society. This could therefore encapsulate potential nonuse 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 willingnesses to pay.

[3] Perfect restoration of the wetland or creation of a perfectly substitutable 'shadow project' wetland, which maintains key features of the original, might have the potential to provide the same nonuse benefits as the original. However, cultural and historical aspects as well as a desire for 'authenticity' may limit the extent to which nonuse 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 wetland or the time frame for restoration.

Evaluation of project, policy or programme options

A combination of quantitative and qualitative research methods is advocated in order to generate a blend of different types of policy relevant information. This applies to both the biophysical assessment of management options and the evaluation of the welfare gains and losses people perceive to be associated with the environmental changes and management responses. The main generic approaches which can form the methodological basis for strategic options appraisal are:

- **stakeholder analysis;**
- **cost-effectiveness analysis;**
- **extended cost-benefit analysis and risk-benefit analysis;**
- **social discourse analysis; and**
- **multi-criteria analysis.**

In summary, comprehensive assessment of ecosystems requires the analyst to undertake the following not necessarily sequential, steps:

1. To determine the causes of ecosystem degradation/loss, in order to improve understanding of socio-economic impacts on ecosystem processes and attributes (e.g. with the aid of the DP-S-I-R auditing framework).

2. Assess the full ecological damage caused by ecosystem quality decline and/or loss.

3. Assess the human welfare significance of such changes, via determination of the changes in the composition of the ecosystem, evaluation of ecosystem functions, provision of potential benefits of these functions in terms of goods and services, and consequent impacts on the well-being of humans who derive use or non-use benefits from such a provision.

4. Formulate practicable indicators of environmental change and sustainable utilisation of ecosystems (within the DP-S-I-R framework).

5. Carry out evaluation analysis using monetary and non-monetary indicators (via a range of methods and techniques, including systems analysis) of alternative ecosystem change scenarios.

6. Assess alternative ecosystem conversion/development and conservation management policies.

7. Present resource managers and policy makers with the relevant policy response options.

A combination of quantitative and qualitative research methods is advocated in order to generate an optimum blend of different types of policy relevant information. This combination of approaches is expected to produce the most useful and meaningful information when compiling indicators. Indicators are commonly understood as quantifiable variables which provide information about changes in for example environmental conditions.

The variable itself may describe an environmental state at a certain point in time and an analysis of these variables over time will provide information about the relevant changes and the rate of change. However, indicators can also be qualitative and descriptive in nature.

The procedure for determining the monetary valuation of environmental change should focus on (1) the identification of the relevant environmental functions that are going to be affected by this change (2) the importance of these functions for sustaining ecosystems and hence human systems (3) who gains or losses from the environmental changes and the socio-cultural and historical contexts that surround particular value gain/loss.

Multi-criteria decision analysis (MCDA) 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 in contexts where a range of, often competing, policy criteria are considered to be socially and politically relevant.

The steps presented in this report towards the development of a holistic integrated framework for environmental indicators are part of an integrated system aiming at the provision of transparent, meaningful and useful information. This system can support and link decision-making at different spatial and time scales with the objective of fostering the protection and sustainable management of natural resources.

Focusing on environmental and social systems and their interactions simultaneously means that the corresponding indicator sets essentially provide the basis for a multi-criteria decision-support framework. Depending on the monitoring scale, in principle the relevant social and environmental effects of decisions can be analysed and evaluated simultaneously. Obviously, this has important consequences for the evaluation of Biodiversity Action Plans (BAPs) in terms of single (function) sites and their role at the landscape ecology level. As discussed, the 'value' of such sites can only by properly addressed when the site is viewed within the larger landscape ecosystem and its contributory value is recognised. The 'natural areas' approach and the 'regional profiles' within the Biodiversity Action Plan championed by English Nature fit the overall systems perspective which is recommended in this report.

Main report

1. Introduction

This report presents the first part of the work commissioned by English Nature, consisting of an overview of the literature relevant to address the project's main objective which is to develop a conceptual framework that represents a standardised integrated approach to support decision-making involving Biodiversity Action Plan (BAP) habitats and species in England. The framework will provide a basis to represent the value and importance of nature conservation. Concepts developed in ecological economics are used to provide a practically useful description of the value and importance of habitats, species and ecosystems.

The research carried out is driven by the need to maintain the functional diversity of ecosystems. In the context of complex decision making which aims to maintain functional and ecological diversity in ecosystems and satisfy multiple stakeholder groups, a range of management options are likely to be available. These options are likely to have different impacts on human and natural systems across different spatial and time scales. The impacts will often be complex, but can be measured with the help of indicators. Capturing the whole range of relevant impacts on natural impacts on natural and human systems within different management options, given the overall goal of sustainable development, will require a combination of environmental, social and economic indicators.

The rest of this section introduces decision making in nature conservation and looks at how decision support systems identify, analyse and evaluate problems. We look at the steps involved in undertaking project and policy appraisal before going on to describe the integrated framework for assessing nature conservation. This takes a systems approach to analyse and evaluate the problem. The relationship between ecosystem functions and environmental values is investigated before we look at the various types of indicators and methods used to describe the importance of nature conservation. We outline the limitations and problems of the approaches and conclude with practical recommendations for implementation of a decision-support system for Biodiversity Action Plans

1.1 Decision support systems

Nature Conservation Assessment Steps

1.1.1 Scoping of the problem and the relevant scale

Here we identify and diagnose the problem, determining its nature, scope and scale. It can be useful to scope the problem in the context of an overall assessment framework, e.g. Driving Forces-Pressures-State-Impact-Response framework (see section 1.3). Such a framework can be used to make explicit how various human activities in a given context and scale relate to the environmental pressures which impact on ecosystem states. These impacts affect environmental changes which in turn result in impacts on human beings and usually some kind of societal response. The societal response feeds back into human activities which in turn affect ecosystems. Scale refers to the spatial and temporal dimensions of the problem. Selection of an appropriate scale is an important element in being able to specify objectives and constraints.

1.1.2 Sustainability issues

Under the sustainability principle there is a requirement for the sustainable management of environmental resources, whether in their pristine state or through sympathetic utilisation, to ensure that the legacy of our current activities does not impose an excessive burden on future generations.

1.1.3 Identify decision evaluation criteria

The specific decision criteria upon which we judge the management objective must be identified. Given the definition of sustainable utilisation used above the pursuit of this objective can be a complex multi-objective problem. Its achievement may require a combination of several evaluation criteria (often traded off against each other) that reflect the different aspects of the objective. In the case of nature conservation this may include criteria such as, maintaining ecosystem health/integrity, preserving critical natural capital, maximising functional value diversity, adoption of the precautionary approach, cost-effectiveness, economic efficiency, social equability, social inclusion, social accountability.

1.1.4 Identify corresponding decision-making calculi/research methods

Once the objectives and decision evaluation criteria have been identified the corresponding decision-making calculi and research methods can be identified. Certain types of criteria will necessitate the use of specific methods, whilst in some case a combination of methods will be required. Examples of such decision-making calculi/research methods include, safe minimum standards, ecological modelling, cost-effectiveness analysis, stakeholder analysis, cost-benefit analysis, multi-criteria analysis. Although all of these approaches may be used when assessing the costs and benefits associated with changes in nature conservation, the roles which they play in the overall assessment can vary greatly.

1.1.5 Identify corresponding information formats/indices

The information format/indices used to indicate the level of environmental change and sustainable utilisation of the ecosystem according to the above calculi are identified (within the DPSIR framework – see Figure 2). These will include, ecosystem thresholds, keystone species, stakeholder identification, financial-economic costs, financial-economic benefits, distribution of financial-economic costs and benefits, degree of public participation

1.1.6 Identify data collection techniques

Depending on the decision-making calculi and information format/indices, the choice of data collection will be determined. Again more than one technique may be necessary. Techniques include, stakeholder mapping, participatory appraisal, focus groups, expert opinion (Delphi Method), species monitoring, market-based data collection techniques, contingent valuation, travel cost, etc.

1.1.7 Evaluate project/policy alternatives

The advantages and disadvantages of the alternatives under consideration need to be evaluated and a ranking of alternatives should result. Whilst the choice of best alternative would seem to be a straightforward proposition, the reality is that in some contexts a clear cut ranking may not emerge. Examination of all the facts, determining whether sufficient information is available and the selection of the best alternative can be more of an art than a precise science.

1.2 Towards an integrated framework for nature conservation assessment

The framework is based on existing approaches to monitoring environmental change and has three distinct features: 1) it adopts a systems approach towards the description of ecosystem integrity (focusing on ecosystem structure, composition and functioning); 2) it combines quantitative and qualitative descriptive indicators of ecosystem integrity (since there often is a lack of rigorous quantitative models to describe the relationships between relevant ecosystem state variables; 3) it can therefore include normative benchmarks to assess the sustainability of natural resource management.

The foundation for the assessment is provided by the Pressure-State-Impact-Response (PSIR) framework, originally developed by the OECD. This co-evolutionary framework relates human activities to environmental pressures that have an impact on ecosystem states and effect environmental change that results in impacts on human beings (society) and usually in some kind of societal response. The societal response feeds back into human activities and subject to various lags back into ecosystems.

The DPSIR framework provides a conceptual and organising backdrop for the contributions of different disciplines to the description and analysis of environmental problems since the social-economic aspects of environmental problems are an integral part of this co-evolutionary framework. The DPSIR framework is an auditing process, not a model. Its main purpose is to make manageable the complexity of environmental problems, for example, in wetland ecosystems and related protection and sustainable management issues. It provides an important starting point on the road towards a common level of *understanding* and *consensus* between researchers, natural resource managers and policy makers about the link between the various driving forces which pose threats to the intrinsic functioning of wetland ecosystems. These pressures include land conversion, agricultural development, hydrological perturbation and pollution, and their consequent impacts on the various interest or stakeholder groups who utilise the goods and services provided by these ecosystems and/or contribute to the pressures on them. Moreover, there are likely to be differences in stakeholders' perceptions of pressures, impacts and environmental values.

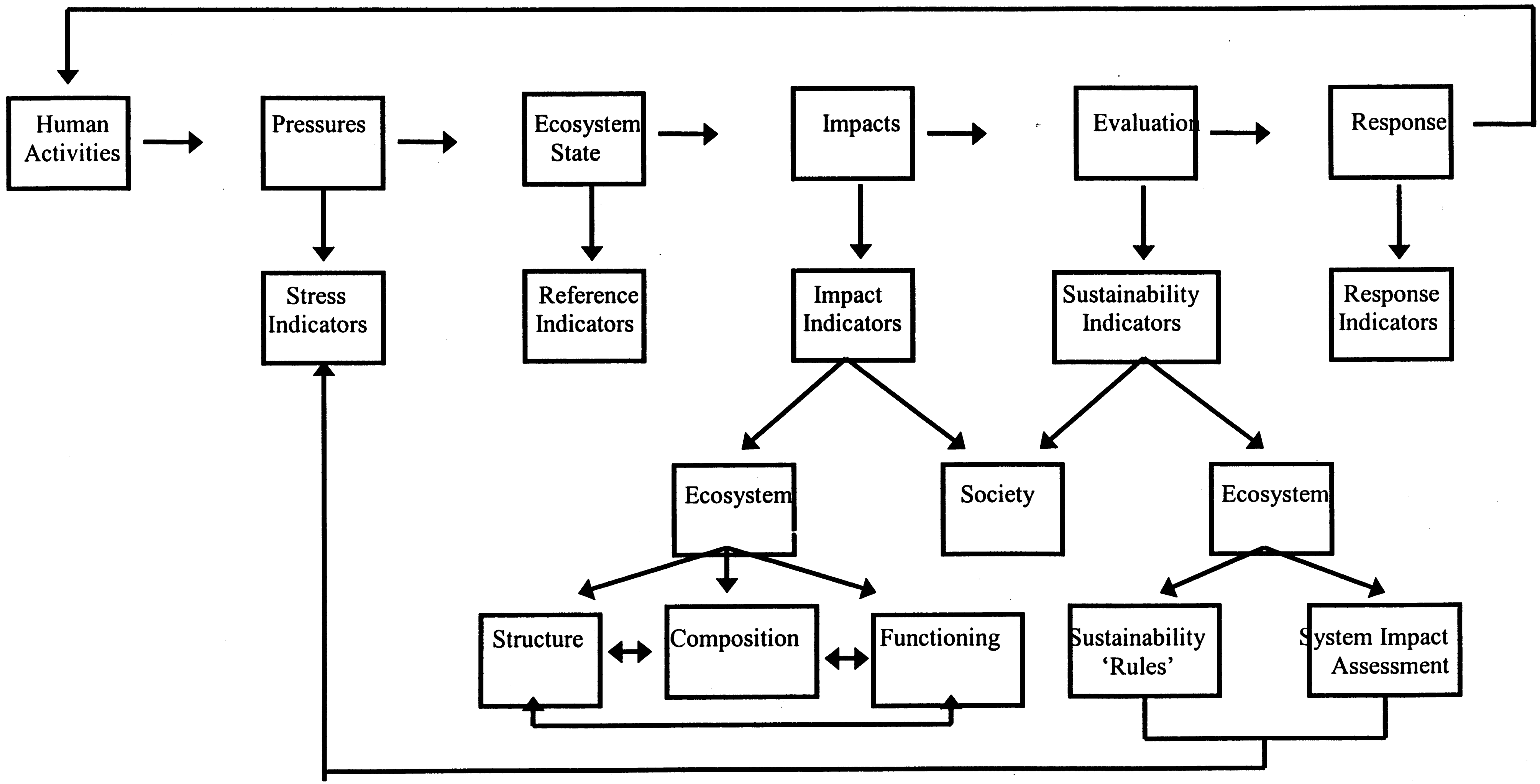
The socio-economic and natural scientific aspects of ecosystem integrity are both an integral part of the model (Figure 1.1). The main aim is to provide a framework in which environmental indicators can be analysed and evaluated vis-à-vis the social context in which they come about. This social context includes the institutional, political, social and spatial scale (local through to international) economic conditions or circumstances in and at which environmental change occurs and is monitored.

The effect of pressures exerted by human activities on ecosystems can be measured by defining the relevant indicator spheres for ecosystem structure, composition and functioning. Management objectives such as sustainable natural resource management and subsequently the impacts of environmental pressures on ecosystem integrity can be assessed in the following two ways:

- First of all, where possible numerical stress and/or effect indicators can be related to ecological, biogeochemical and hydrological benchmarks or sustainability rules derived from natural scientific literature to determine ecosystem sustainability. For example, emission loads into an ecosystem can be compared to the assimilative capacity of the specific ecosystem.

- Secondly, ecosystem integrity can be assessed by investigating the impact of environmental pressures on structural, compositional and functional ecosystem changes and the impact of these changes on each other. This will likely be a more qualitative descriptive analysis. For example, human induced stress may alter ecosystem structure or composition. The impact of this change in ecosystem structure or composition on general ecosystem functioning has to be investigated to assess ecosystem integrity (see section 3.3. for detailed discussion).

Figure 1.1: General framework for the derivation of environmental indicators to evaluate the sustainability of natural resource management



2. A systems approach to nature conservation assessment

2.1 Introduction

Recent advances in the development of ecological economic models and theory all seem to stress the importance of the overall system, as opposed to individual components of that system. This points to another dimension of total environmental value, the value of the system itself. The economy and the environment are now jointly determined systems linked in a process of co-evolution, with the scale of economic activity exerting significant environmental pressure. The dynamics of the jointly determined system are characterised by discontinuous change around poorly understood critical threshold values. But under the stress and shock of change the joint systems exhibit resilience, i.e. the ability of the system to maintain its self-organisation while suffering stress and shock. This resilience capacity is however related more to overall system configuration and stability properties than it is to the stability of individual resources. This has important implications for the assessment of individual conservation sites which within their own boundaries may service only a single function or host a single 'important' species. The 'value' of such sites can only by properly addressed when the site is viewed within the larger landscape ecosystem and its contributory value is recognised (see section 3). The 'natural areas' approach and the 'regional profiles' within the Biodiversity Action Plan championed by English Nature fit the overall systems perspective which is recommended in this report.

2.2 Key ecological concepts

An important extension of the original conceptual assessment framework is that the revised framework explicitly recognises the role of sustainability indicators. This is done using the concept of *ecosystem integrity*, a concept which has recently grown in popularity. Ecosystems are defined as 'spatially explicit units of the Earth that include all of the organisms, along with all components of the abiotic environment within its boundaries' (Likens, 1992). The boundaries of an ecosystem should be defined so as to measure, monitor and manipulate the properties and processes that are of interest. Nevertheless, whatever the boundary or spatial scale looked at, ecosystems are always open with respect to the inputs and outputs of energy and matter. *Ecosystem Integrity* is defined as the maintenance of system components, interactions among them and the resultant behaviour or dynamic of the system (King, 1993). The 'integrity' of an ecosystem is more than its capacity to maintain autonomous functioning (its health); it also relates to the retention of 'amount of diversity'. i.e. the species and interrelationships that have survived over time at the landscape level.

Sustainability has shown to be hard to operationalise and often appears to be an empty concept. Moreover, the fact that nowadays simple descriptive environmental indicators are often referred to as sustainability indicators further adds to the confusion. Sustainability can only be defined, ultimately, at the level of the interaction between the entire complex of human systems and all directly and indirectly implicated environmental systems (Clayton and Radcliffe, 1996). To understand sustainability therefore requires understanding of the behaviour of systems in general and of human and environmental systems in particular. Hence the holistic approach adopted in this assessment process.

An attempt will be made to incorporate the concept of ecosystem integrity in the general indicator framework. In the past, other '*sustainability*' concepts have been developed as well such as '*ecosystem health*' (Costanza *et al.*, 1992) or '*ecosystem resilience*' (Holling, 1986). The former is defined as a system free of distress syndrome, while the latter refers to a system's ability to maintain its structure and pattern of behaviour in the presence of stress. Ecosystem integrity resembles the above mentioned concepts in that they all refer to, implicitly or explicitly, a certain minimum structural system composition required for the overall functioning of ecosystems. A lot of subsequent discussion about the usefulness of these different concepts has been dominated by semantics. For our purposes, the most important characteristic of these concepts that has been taken on board here is that they adopt a systems approach to the analysis of complex natural resources.

Ecosystem integrity

The concept of ecosystem integrity covers various natural scientific disciplines with systems science playing an important synthesising background role. The concept of ecosystem integrity was first introduced by Leopold (1939) and has since been defined in various ways in ecological and biological literature (e.g. Cairns, 1977; Karr and Dudley, 1981; Karr *et al.*, 1986; Kay, 1989). Over the past decade, it has nevertheless grown in influence, suggesting a transition in societal understanding of our relationship to the physical, chemical and biological environment (Karr, 1993). Although there exists a relatively well developed body of literature about ecosystem integrity, especially for large aquatic ecosystems (Edwards and Regier, 1990; Spigarelli, 1990), there seems to be a lack of general agreement amongst experts about the meaning and relevance of the concept and the measures needed to index it (Steedman and Haider, 1993).

However defined or described, systems theory provides an important analytical framework for the structure and functioning of nature and environment given its emphasis on the identification and description of the connections between objects and events as well as the objects and events themselves. Formally, a system is a set of components that interact with each other. Changes in one component will induce changes in another, which may in turn induce change in a third component. Any one interaction of this kind is causal and directional, while many such interactions can be linked together in chains of cause and effect relationships.[1] King (1993) argues that the description of a system simultaneously involves both structure and function: what are the components, how are they connected and how do they operate together? According to King, system integrity thus implies the integrity of both system structure and function[2], a maintenance of system components, interactions among them and the resultant behaviour or dynamic of the system.

Given the emphasis on both system components and the interactions between them, this implies that the loss of system integrity has to be evaluated in the light of both structure and function or processes. In the context of biological diversity (biodiversity), Franklin *et al.* (1981) distinguish a third criterion or attribute to identify and describe ecosystems besides

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[1] A given component can often, in practice, operate both in a control function (causing change in another) and in a dependent function (being changed by another). Chains of cause and effect relationships can intersect themselves. This means that a component can start a sequence of cause and effects that eventually loops back, so that each of the components in the loop indirectly influences itself.

[2] It is important to note that ecosystem function refers here to the actual operation of ecosystems or ecosystem processes (*e.g.* nutrient recycling), not their role as perceived for example by economists (*e.g.* life support functions provided by ecosystems).

structure and function, namely ecosystem composition. Composition refers to the identity and variety of elements in a system whereas structure refers to the physical organisation of a system. The loss of a single system component such as the loss of a single species or population (resulting in a change of ecosystem composition and/or structure) or a change in interaction (processes) does not necessarily imply loss of system integrity. Many systems, including ecological systems, appear to be resilient to alteration of structure. Whole system function is maintained despite the structural change. Ecosystem components may perform equivalent functions and loss of one or more may produce very little change in whole system function (Harcombe, 1977; Foster *et al.*, 1980; Rapport *et al.*, 1985; O'Neill *et al.*, 1986; Vitousek, 1986).[3] This is also called functional redundancy (King, 1993).

Ecosystem resilience

In the definitions presented in the literature, stability and adaptation seem to be two key characteristics of ecosystem integrity. Here we will call this ecosystem resilience, that is, a system's capability to maintain stability in the presence of external (often human induced) disturbances. System states with greater stability are, by that very virtue, displaced less quickly than their less stable counterparts (Clayton and Radcliffe, 1996). Similarly, sub-components of systems, which are more stable will tend to exist longer. All biological and ecological systems have a degree of resilience. They will tolerate a certain level of stress or depredation, while maintaining the capacity to recover. Even if individual elements of a system are destroyed, these elements can often be restored provided that the essential network of relationships that constitutes the system remains. In general, the more complex an ecology, and the more interlocking feedback systems there are, the more robust and better able to resist change the system appears to be.[4]

An important feature of dynamic systems, such as most ecological systems, is that they can be ordered and stable. This stability is a function of the interaction of individual elements in the system counterparts (Clayton and Radcliffe, 1996). An ecology, for example, can be maintained in a stable state by a dynamic interaction between the species that constitute that ecological system. Or, in society the aggregate behaviour of companies, consumers and markets can be stable even though the individual buying and selling decisions of the individuals that constitute the community cannot be predicted.

2.2.1 Ecological versus biological syem integrity

In the ecosystem integrity literature, a distinction between '*ecological*' and '*biological*' system integrity can be found, called the 'process-functional' and the 'population-

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[3] Primary productivity or nutrient cycling may remain relatively constant while species composition changes (Harcombe, 1977; Rapport *et al.*, 1985) or dominant species are removed (Foster *et al.*, 1980).

[4] The global ecology maintains, for example, a dynamic homeostasis via factors that contribute inertia and negative feedback loops that will tend to compensate for and hence resist change. This is somewhat counter intuitive, because there often appear to be more ways to break complex systems than simpler ones. In practice, however, complex systems tend to exhibit greater stability. One reason for this may be that some complex systems, such as biological systems, employ multiple rather than single pathways of control. Systems that employ multiple determinacy tend to have enhanced chances of survival. More fundamental, complex systems generally evolve from simpler systems rather than arriving in a fully formed state. The incremental complexity will normally develop only if it confers a net adaptive or selective advantage to the 'parent' system. Part of this advantage may accrue precisely because the complexity makes the system more robust. Even if this is not the case, the more complex system will only be able to survive if it is highly robust, because it will inevitably suffer many interferences during the course of its existence. This is true to a greater extent for complex systems than simple systems, because there are more parts to be interfered with, so the stability is a natural characteristic of these systems (Clayton and Radcliffe, 1996).

community' approach respectively by O'Neill *et al.* (1986). In the former approach, the system of biotic population interactions is emphasised and the abiotic physical environment is seen as external to the system, while in the latter approach the physical environment is an integral part of the system of matter and energy transformations through biota and environment. This distinction not only depends upon differences in perspectives regarding scale, but also on the distinction in systems science between open and closed systems.

In systems science, the distinction between open and closed systems depends upon thermodynamics (Clayton and Radcliffe, 1996). Closed systems have unchanging components. They will eventually arrive at an equilibrium and tend to move towards a state of higher entropy. This means that closed systems usually reach a point at which no further change is possible for that system. Open systems exchange flows with their environment. These flows can consist of materials, energy or information. Open systems can reach a steady state which depends upon their being able to maintain continuous exchanges with their environment. This is what allows open systems to create and maintain a state of low entropy. This means that some open systems can maintain their integrity as systems, although this must always be at the expense of an increase of entropy elsewhere.

Most living systems are open systems. The environments in which living systems live are of course themselves never completely stable, so living systems must try to obtain reasonably stable flows from sources that can be changing over time. This means that living systems and ecological hierarchies of living systems must have processes of communication and control so that they can monitor and respond to and in that way resist the perturbations of a real-life environment.[5]

2.2.2 Ecosystem scales and hierarchies

Scale and hierarchy are essential aspects to the concept of ecosystem integrity (King, 1993). Scale refers to the spatial and temporal dimensions of the system, while hierarchy is a ranked ordering of interactions or levels of organisation. Scale includes both 'grain' and 'extent' (Turner and Gardner, 1991). Grain is the finest level of temporal or spatial resolution in an observation set, while extent is the areal expanse or the length of time over which observations with a particular grain are made. Obviously, the choice of scale determines the description of the ecosystem. For the purpose of ecosystem integrity assessment, a balance should be sought between larger and finer scale measurement. Although ecosystem integrity refers to a system's wholeness, suggesting the measurement of whole system properties, holistic larger scale observations may disguise relevant finer scale variability. Hence, depending on the policy issue at hand, larger scale measurements may have to be complemented by finer scale measurements.

Ideally, environmental impacts are measured at multiple levels of organisation and at multiple spatial and temporal scales, because the effects of environmental pressures will be expressed in different ways at different but highly interdependent levels of biotic and abiotic organisation. For the purpose of compiling environmental indicators, this means that for the relevant ecosystem attributes, such as structure, composition and functioning, the relevant level(s) of organisation has to be determined for the specific policy question at hand. Concentrating on specific ecosystem attributes in isolation may provide an incomplete and

[5] It is important to note that this does not have to be in any sense a conscious process. Effective control in a changing environment requires that systems have control mechanisms with a variety of response that can match the variety of the environmental information (Clayton and Radcliffe, 1996).

possibly incorrect picture of the environmental state or condition the policy maker is interested in. The choice of level of organisation depends on the available information about relationships between different levels. Inferences about species distributions can for example be drawn from inventories of vegetation. Remote sensing can effectively monitor the availability of habitats over broad geographic areas (Noss, 1990). The importance of higher order constraints (nested hierarchies) does not suggest that monitoring and assessment be limited to higher levels. Relevant information about possible ecosystem tolerance levels, thresholds or feedback loops may be found at lower levels in a hierarchy. Or lower hierarchical levels may contain the details of interest to conservationists.

Finally, although we have discussed ecosystem integrity so far in ecological and biological terms, it has strong social-economic, cultural and ethical dimensions as well. Nowadays, human beings are more widely considered to be part of natural systems, exerting considerable pressures on natural systems, form natural systems (to some extent) by their decisions and hence implicitly or explicitly value natural systems change. In a systems approach, we consider human beings and the interactions between human beings (social-economic activities) and their natural environment as much an integral part of the system's environment as ecology and biology. They are in effect all essential dimensions of sustainable development.

3. Ecosystem functions and their value to society in making the link

3.1 Introduction

In the next two sections we discuss how we might assess the concept of ecosystem integrity in ecosystem management policies vis-à-vis human activities and the pressures exerted by these activities. As we shall see we can relate ecosystem integrity in an operational way, as some minimum configuration necessary for the operation of an ecosystem, to concepts such as 'primary value' and 'safe minimum standards'.

Ecologists refer to ecosystem functioning variously as the habitat, biological or system properties or processes of ecosystems. There are a variety of ecosystem processes that are critical to the sustained functioning of ecosystems. These include the conversion of solar energy to chemical energy by plants, the cycling of elements such as carbon, nitrogen and oxygen, and the flow and transfer of water.

Ecosystem goods and services represent the benefits human populations derive from sustained ecosystem functioning. Capital is considered to be a stock of materials or information that generates (sometimes in conjunction with services from other capital stocks) a flow of services that enhance human welfare. Human use of these flows of services may deplete the capital stock. We can distinguish between three broad types of capital. **Man-made capital** (factories, roads, houses, etc.), which can be increased or decreased at our discretion; **Critical natural capital** (ozone layer, global climate, biodiversity, wilderness, etc.), which comprises natural assets essential to life that cannot be replaced or substituted by man-made capital (note that there are natural assets such as elements of biodiversity that whilst not essential to life support may still have a life support role in terms of making life 'tolerable'); **Other natural capital**, which includes renewable natural resources and some finite mineral resources that can be wholly or partly replenished or substituted by man-made capital.

Natural capital stock and the associated flows of materials and services combine with manufactured and human (intellectual) capital services to enhance human welfare. While it is possible to imagine generating human welfare without natural capital and ecosystem services, it is often considered that the general class of natural capital is essential to human welfare. Natural capital and man made capital are thus argued to be largely complements rather than substitutes. Zero natural capital implies zero human welfare because non-natural capital cannot feasibly be substituted for natural capital. As such the total value of natural capital to human welfare is infinite. However, in many real world circumstances the policy debate concerns 'marginal' changes in the ecosystem and natural capital stock. It is thus in this context that it is meaningful to ask how changes in the quantity or quality of various types of natural capital and ecosystem services impact on human welfare.

A number of ecosystem goods and services can be valued in economic terms, while others cannot because of uncertainty and complexity conditions. Ecosystems provide a wide range of goods and services of significant value to society - storm and pollution buffering function, flood alleviation, recreation and aesthetic services, etc. We can therefore conceive of 'valuing' an ecosystem as essentially valuing the characteristics of a system, and we can capture these values in our economic value framework. But since it is the case that the component parts of a system are contingent on the existence and continued proper functioning of the whole, then putting an aggregate value on ecosystems is quite a complicated matter.

3.1.1 Environmental function diversity evaluation

In order to consider ecosystem and component valuation we use an interdisciplinary analytical framework that has at its core a conceptual model, based on the concept of functional value diversity. This model links ecosystem processes and functions with outputs of goods and services, which can then be assigned monetary economic and/or other values (Figure3.1).

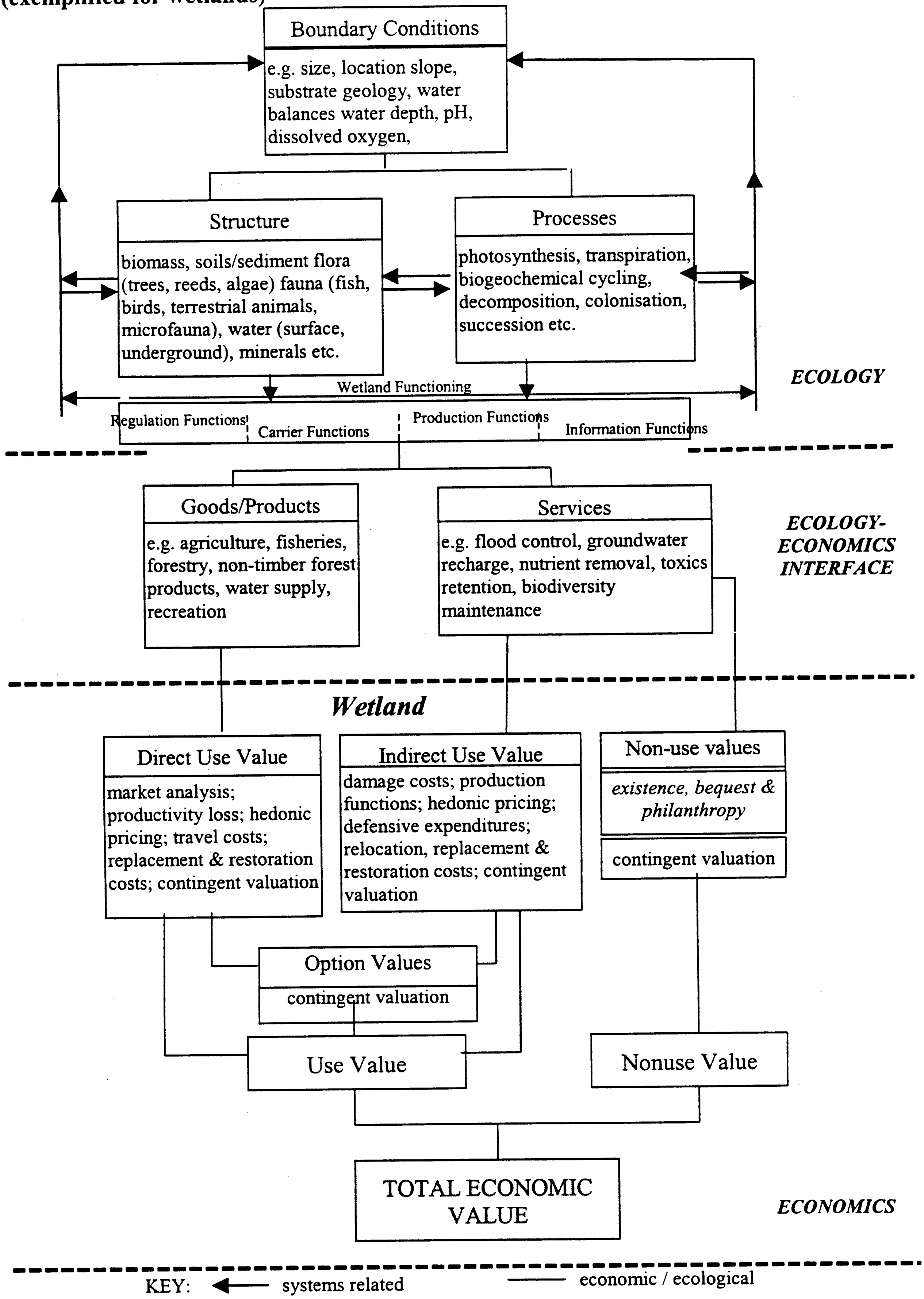
Functional diversity from an ecological stance can be defined as a variety of different responses to environmental change, in particular the variety of spatial and temporal scales with which organisms react to each other and to the environment. The policy objective of maximum diversity maintenance serves to ensure the maximum amount of functional capacity and associated functional value in terms of goods and services provision. The model requires the practical coupling of economic, hydrogeomorphological and ecological models.

Social scientists on the other hand refer to functional diversity in terms of maintaining as many functions of an ecosystem as possible. Their interest lies more in functional value diversity. The diversity concept encourages analysts to take a wider 'landscape ecology' perspective in terms of examining changes in large scale (land-use level) ecological processes, together with the relevant socio-economic driving forces causing ecosystem loss. The focus is then on the ability of interdependent ecological-economic systems to maintain functionality under a range of stress and shock conditions i.e., it relates back to the concepts of ecosystem integrity and resilience. The value of ecosystem functions then, should be assessed not just at the local scale level, but at the landscape ecology level, e.g. catchments.

The **function** concept is an important element in this model. Ecologists consider the function concept in terms of the capacity and workings of ecosystem processes (functioning process), whereas *functions* are defined in socio-economic terms, as relating to the actual provision of goods and services that satisfy human needs and wants, both physiological and psychological. The satisfaction of these needs and wants and the performance of many human activities, depend on certain environmental conditions.

It is argued that translating these environmental capacity conditions into functional provisions of the natural environment, instead of the narrower concept of natural resources provides a useful analytical framework for measuring environmental health and quality of life. Such an approach is behind the methodology proposed for the assessment of environmental capital, known as the Environmental Capital Approach (CAG & LUC, 1997). This approach looks at characterising the services and benefits that an environmental area or asset provides, and looks at maintaining these services, rather than preserving the asset itself. Whilst not directly concerned with analysing trade-offs and relative values of one service for another, the approach is useful as an appraisal auditing framework.

Figure 3.1 From Ecosystem Structure and Processes to Ecosystem Functions and Values (exemplified for wetlands)



Maltby (1999) provides further arguments that support a functional approach to developing a decision support system for ecosystem conservation and management:

1. It should allow more efficient use of scarce resources by determining such relationships as the compatibility and intensity of land use activities with functioning, the capacity of ecosystem to tolerate impacts, and their resilience to human disturbance.

2. 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.

3. Ecosystem dynamics are more translatable into economic terms, which are usually more understandable to the public and politicians than more ethical and scientific arguments.

4. 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.

5. It allows scope for policy innovation.

6. Assessment of ecosystem functioning should lead to more effective environmental protection since the ability to target more precisely the systems responsible for particular benefits is possible. Two dimensions to this are: optimising the use of limited financial resources; identifying priority areas for protection, rehabilitation or restoration since there will exist forecasts of likely success.

Another important facet of this approach is the derivation of scientifically valid and practical indicators of environmental change and sustainability.

De Groot (1992), distinguishes between four function categories (see table 3.1):

1. Regulation functions: relating to the capacity of natural and semi-natural ecosystems to regulate essential ecological processes and life support systems which, in turn, contributes to the maintenance of a healthy environment by providing clean air, water and soil.

2. Carrier functions: natural and semi-natural ecosystems provide space and suitable substrate or medium for many human activities such as habitation, cultivation and recreation.

3. Production functions: nature provides many resources, ranging from food and raw materials for industrial use to energy resources and genetic material.

4. Information functions: natural ecosystems contribute to the maintenance of mental health by providing opportunities for reflection, spiritual enrichment, cognitive development and aesthetic experience. In addition there are probably many unknown goods and services (functions) which are not yet recognised, but which may have considerable (potential) benefits to human society.

Table 3.1 Life support functions of the natural environment

Regulation Functions	Carrier Functions Providing space and a suitable substrate inter alia for:	Production Functions	Information Functions
Protection against harmful cosmic influences Climate regulation Watershed protection and water catchment Erosion prevention and soil protection Storage and recycling of industrial and human waste Storage and recycling of organic matter and mineral nutrients Maintenance of biological and genetic diversity Biological control Providing a migratory, nursery and feeding habitat	Habitation Agriculture, forestry, fishery, aquaculture Industry Engineering projects such as dams, roads Recreation Nature Protection	Oxygen Food, drinking water, and nutrition Water for industry, households, etc Clothing and fabric materials Building, construction and manufacturing materials Energy and fuel Minerals Medical resources Biochemical resources Genetic resources Ornamental resources	Aesthetic information Spiritual and religious information Cultural and artistic inspiration Educational and scientific information Potential information

None of these environmental functions can take place in isolation. They are the result of the dynamic and evolving structures and functions of their total ecological sub-system, and the fact that the socio-economic values of environmental functions and ecological sub-systems are directly connected to their physical, chemical, and biological role in the overall global system.

The capacity of a given ecosystem to provide goods and services depends on its environmental characteristics (natural processes and components) which set the boundary conditions. Ideally, a matrix should be developed showing the relation between environmental characteristics and environmental functions (see Table 3.2). This should focus on those characteristics which are important as parameters or criteria for assessing or evaluating the qualitative and/or quantitative capacity of a given area or ecosystem to provide certain functions. The large number of natural processes and components which influence the functions provided by ecosystems have been grouped by De Groot (1992) into 9 main categories: Bedrock characteristics and geological processes; atmospheric properties and climatological processes; Geomorphological processes and properties; Hydrological processes and properties; Soil processes and properties; Vegetation and habitat characteristics; Species-properties and population dynamics; Life-community properties and food chain interactions; Integrated ecosystem characteristics.

Table 3.2 Relationship between Environmental Functions and Environmental Characteristics

Environmental Characteristics	Environmental Functions			
	Regulation	Carrier	Production	Information
Bedrock/Geology (e.g. lithology)				
Atmosphere and Climate (e.g. air quality)				
Geomorphology/Relief (e.g. sedim/erosion)				
Hydrology (e.g. water quality)				
Vegetation (e.g. structure)				
Flora and Fauna (e.g. species diversity)				
Life Community (e.g. food chain interactions)				
Soil (e.g. fertility)				
Ecosystem (e.g. naturalness)				

3.1.2 Ecosystem processes and their relationship to specific ecosystem components

Before considering the valuation of ecosystem functions let us first consider a number of key issues and ecological principles relating to the functioning of ecosystems and the assignment of values to ecosystem structure and functions.

1. Spatial and Temporal Scale - Ecological processes operate over a range of spatial and temporal scales; scales of time or space appropriate for the study of management of one process may not be suitable for other processes.

2. Ecosystem Function Depends on Structure, Complexity and Diversity - Biological diversity underlies the complexity of ecosystem function. One question frequently asked is how many species are needed to maintain key ecosystem functions? This question however assumes that there is a predictable overlay between species diversity and functional diversity, that all the individual organismal activities that comprise overall ecosystem function are known, and that ecosystems do not change in ways that may influence which species are best able to carry out key functions. In fact, these assumptions are false. Several species can play similar roles in a functional context, thus the notion of ‘redundancy’ in understanding the value of diversity (see later).

Biological diversity also provides for both stability (resistance) for and recovery from disturbances that disrupt important ecosystem processes.

Diversity, furthermore, provides long term capacity for adaptation, as well as being a sensitive indicator of environmental change. It is important here to acknowledge that the contribution of diversity to ecosystem functions is not merely a function of the number of species. Their identity is also important.

3. Ecosystems are Dynamic in Space and Time - Change is the normal course of events for ecosystems. Natural or human induced disturbance on landscapes creates a patchwork mosaic, and the resulting changes initiated within each patch are influenced by the pattern and behaviour of surrounding patches. This landscape variation influences ecosystem functioning at large spatial scales.

4. Uncertainty and Surprise are Inevitable - There is much that is not understood about ecosystems. Some of that ignorance will yield to increased knowledge, but the complexity and interactions of non-linear processes means that certain elements of ecosystem function will always be difficult to predict and that surprises in ecosystem behaviour are inevitable.

3.2 Functions and categories of value: implications for environmental valuation

3.2.1 Introduction

The key to valuing a change in an ecosystem function is establishing the link between that function and some service flow valued by people. If that link can be established, then the concept of derived demand can be applied. The value of a change in an ecosystem function can be derived from the change in the value of the ecosystem service flow it supports.

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

However, the multifunctional characteristic of ecosystems makes comprehensive estimation of every function and linkages between them a formidable task. Barbier (1993) thus considers three broad approaches to ecosystem change valuation, the use of each depending on the type of threat and the potential means of conservation. The three approaches are:

- impact analysis, assessing the damage from a particular impact;
- partial valuation, considering specific functions or areas of the ecosystem;
- total valuation, estimating the full present value of an ecosystem.

3.2.2 Categories of value

In instrumentally valuing a resource such as an ecosystem, the total economic value (TEV) can be usefully broken down into a number of categories as shown in figure 3.2. The initial distinction is between use value and non-use value. Use value involves some interaction with the resource, either directly or indirectly:

1. Indirect use value derives from services provided by the ecosystem. This might for example include the removal of nutrients, providing cleaner water to those downstream, or the prevention of downstream flooding.

2. Direct use value, on the other hand, involves interaction with the ecosystem itself rather than via the services it provides. It may be consumptive use such as the harvesting of reeds or fish, or it may be non-consumptive such as with some recreational and educational activities. There is also the possibility of deriving value from 'distant use' through media such as television or magazines, although whether or not this type of value is actually a use value, and to what extent it can be attributed to the ecosystem involved, is unclear.

Non-use value is associated with benefits derived simply from the knowledge that a resource, such as an individual species or an entire ecosystem, is maintained. It is by definition not associated with any use of the resource or tangible benefit derived from it, although users of a resource might also attribute non-use value to it. Non-use value is closely linked to ethical concerns, often being linked to altruistic preferences, although for some analysts it stems ultimately from self-interest. It can be split into three basic components, although these may overlap depending upon exact definitions.

3. Existence value can be derived simply from the satisfaction of knowing that some feature of the environment continues to exist, whether or not this might also benefit others. This value notion has been interpreted in a number of ways and seems to straddle the instrumental/intrinsic value divide.

4. Bequest value is associated with the knowledge that a resource will be passed on to descendants to maintain the opportunity for them to enjoy it in the future.

5. Philanthropic value is associated with the satisfaction from ensuring resources are available to contemporaries of the current generation.

Finally, two categories not associated with the initial distinction between use values and non-use value include:

6. Option value, in which an individual derives benefit from ensuring that a resource will be available for use in the future. In this sense it is a form of use value, although it can be regarded as a form of insurance to provide for possible future but not current use.

7. Quasi-option value is associated with the potential benefits of awaiting improved information before giving up the option to preserve a resource for future use. It suggests a value in particular of avoiding irreversible damage that might prove to have been unwarranted in the light of further information. An example of an option value is in bio-prospecting, where biodiversity may be maintained on the off-chance

that it might in the future be the source of important new medicinal drugs. It has been suggested that option value is less a distinct category of total value than the difference between an ex-ante perspective yielding 'option price' (consumer surplus plus option value) and an ex-post perspective giving expected consumer surplus, as a measure of value.

3.2.3 Ecological structure and functions of biodiversity as elements of total economic value[1]

The above functional approach to assessing the value of ecosystem functioning and functions is fine for multi-purpose ecological systems. However problems arise if we instead consider the case of single function sites. This applies especially to the case of biodiversity maintenance, where we look at individual habitats and species. Such cases are now discussed within the overall context of ecological structure and functions of biodiversity and their relevance for man's existence.

Biodiversity protection generally requires marginal decisions and so the focus on integrating the ecological importance of biodiversity into TEV focuses on the marginal value of biodiversity. In this context we focus on the ability of natural assets to touch on the welfare position of individuals, directly as an argument in their utility position and/or via the production process. The concept of Total Economic Value can thus be traced back to two fundamental categories of value: production and individual values.

Production Values of biodiversity are arguments in the production and cost functions of market allocated goods. These production inputs affect individual welfare via changing prices of goods or other inputs, e.g. use of ecosystems for agriculture and forestry production. Individual Values on the other hand, are a direct argument of individual utility functions. These include recreational and aesthetic values, as well as passive use, non-use or existence values.

Recently however, these two categories of value have been supplemented with another category, which considers the ecological importance of biodiversity, by describing the ecological-functional role of biodiversity in natural systems. Included here are those services of biological resources that stabilise the ecological system and perform a protective and supportive function for the economic system. The recently developed approaches for considering the ecological functions of biodiversity include the following (somewhat overlapping) categories of values under this umbrella:

- inherent value – describes those services without which there would not be the goods and services provided by the system (Farnworth *et al.*, 1981);
- contributory value – considers the economic-ecological importance of species diversity, such that even species not useful for human use are important since they contribute to increases in diversity which contributes to the generation of more species (Norton, 1986);
- indirect use value – is related to the support and protection provided to economic activity by regulatory environmental services (Barbier, 1994);

[1] This section draws heavily on material from Fromm, 1999

- primary value – incorporates the fact that the existence of the ecosystem structure is prior to the range of function/service values (Turner and Pearce, 1993);
- infrastructure value – relates to a minimum level of ecosystem 'infrastructure' as a contributor to its total value (Green *et al*, 1994; Costanza *et al*, 1997).

As far as implementing the ecological relevance of biodiversity into the total economic value framework is concerned, these categories consider services of biodiversity which are different in content as well as being terminologically different. The differences can be traced to specific complementary relationships of which there are three particular aspects:

1. A **complementary relationship** of species in their habitat – A number of studies have been undertaken on the quantification of the production and individual values of species. The production value of species diversity focuses mainly on the use of species for the development of pharmaceutical products. The individual value of species looks instead at their recreational and existence values.

   The importance of species diversity is that the co-existence of species within a habitat is defined by complex relationships of interaction and interdependence. The 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. Here the diversity of plant species and micro-organisms are intermediate goods for the productive and individual use of animals and plants and so their loss is still economically relevant, even though it may not be directly useful. An example of such 'contributory value' is in the case of the productive use of wild species for the preservation of the resistance of cultivated plants.

   Economic valuations of single species will be incomplete if their 'contributory value' is not taken into account. Inclusion of such values, besides the production and individual values, points to the limits of substitutability of species. The limited substitutability of species results, from an ecological point of view, from the fact that every species performs very specific duties within ecological systems.

   Taking into account 'contributory values' requires complete knowledge of the ecological interrelationships of single species to directly beneficial species. The relevance of the 'contributory values' therefore lies more in a qualitative evaluation in the sense that it reveals the complementary relationship of species and is often associated with a more precautionary approach to conservation.

2. **Ecosystem functions**. The loss of certain species will have only minimal effect on ecosystem functions, since the functionality of ecosystems depends on a limited number of biotic and physical processes. These processes are directed by different groups of species with complementary functions, known as "keystone species".

   This does not mean that ecosystems contain a large number of ecologically redundant species. Species that may seem redundant under certain environmental conditions can become "keystone species" under different conditions. So long as specific species can substitute each other under the changing conditions, the balancing process within and between ecosystems remains intact.

Whether there is a possible critical level of species diversity, which is essential for the functionality of ecosystems, can be looked at from an economic point of view as the search for substitutability. It involves reducing, from an ecological point of view, the relevant functions of species to a finite number. However since knowledge of the functional substitutability of ecological structures is highly imperfect, strictly speaking defining a critical (essential) ecosystem structure is impossible. Having said all this, there is general agreement amongst ecologists that with the reduction of species the possibilities for substitutions necessary for the preservation of functions under changing environmental conditions diminish. The probability of a transition of the ecosystem into different states increases. This is the fundamental importance of functional diversity in ecosystems.

We can interpret the structures and functions of ecosystems in economic terms as the difference between stocks (structural components) and flows (functions). The structural components function as assets, whose combinations result in certain functions of ecosystems. As such single species can be considered as an input for the production of ecological functions, rather than as an output of biodiversity (corresponding to production and individual values). The structural components are preliminary to the ecological functions (primary value – see later). A certain critical ecosystem structure is required if functionality is to be guaranteed.

3. **The complementary relationship of the ecological functions of ecosystems and the contribution of the services of ecosystems to human welfare** – This is the subject of most of the value typology work for defining the ecological relevance of biodiversity. Ecological functions provide the basis for those services then directly used and valued by individuals and the complementary nature of the relationship is what is behind the idea of primary and secondary values of ecological systems. The idea here originates from the work of Farnworth et al (1981) who drew attention to the fact that ecological services are inherently connected to the integrity of natural systems and embody the totality of structure and functioning of the system. Ecological systems thus posses 'inherent values', as values that support all other values. Gren et al (1994) developed this approach into the concepts of 'primary' and 'secondary' values. Primary value (PV) consists of the system characteristics upon which all ecological functions depend (Turner and Pearce, 1993). It describes the self organising capacity of the system, including its dynamic evolutionary processes and its capability to absorb external disturbances. The primary value and continuous preservation of the ecosystem health is the source of functions that have secondary value. These secondary functions and values depend on the continued 'health', existence, operation, and maintenance of the ecosystem as a whole. The primary value notion is related to the fact that the system holds everything together (and is thus also referred to as a 'glue' value) and as such has, in principle, instrumental value. As an example, and in terms of stocks and flows, we can consider the ecosystem Primary Value as the minimum stock (critical natural capital) of habitats at the landscape level required to maintain the landscape structural diversity; and Secondary values as a renewable flow of goods and services generated by the maintained ecosystem.

The contribution of the concept of "indirect use values" (Barber 1994) is that it clarifies part of the linkage between ecological functions and human welfare.

The conventional total economic value restricted to the individual and production values of biodiversity, contains no value component that gives credit to these connections and, therefore, remains incomplete.

Turning back to the discussion of complementary relationships the conclusion from this is that the total value of biodiversity is infinite. As a basis of human life, it is indispensable under realistic technological and economic conditions. **Marginal decisions or apparently marginal decisions as perceived by different stakeholders, are the stuff of the real world political economy, and therefore need to be considered. The problem is that knowledge about the consequences of infringements on biodiversity is incomplete. There is an unbridgeable gap in our knowledge about ecosystem interrelationships and regularities. The benefits of biodiversity protection will often only be discovered, once they have been disturbed or lost.**

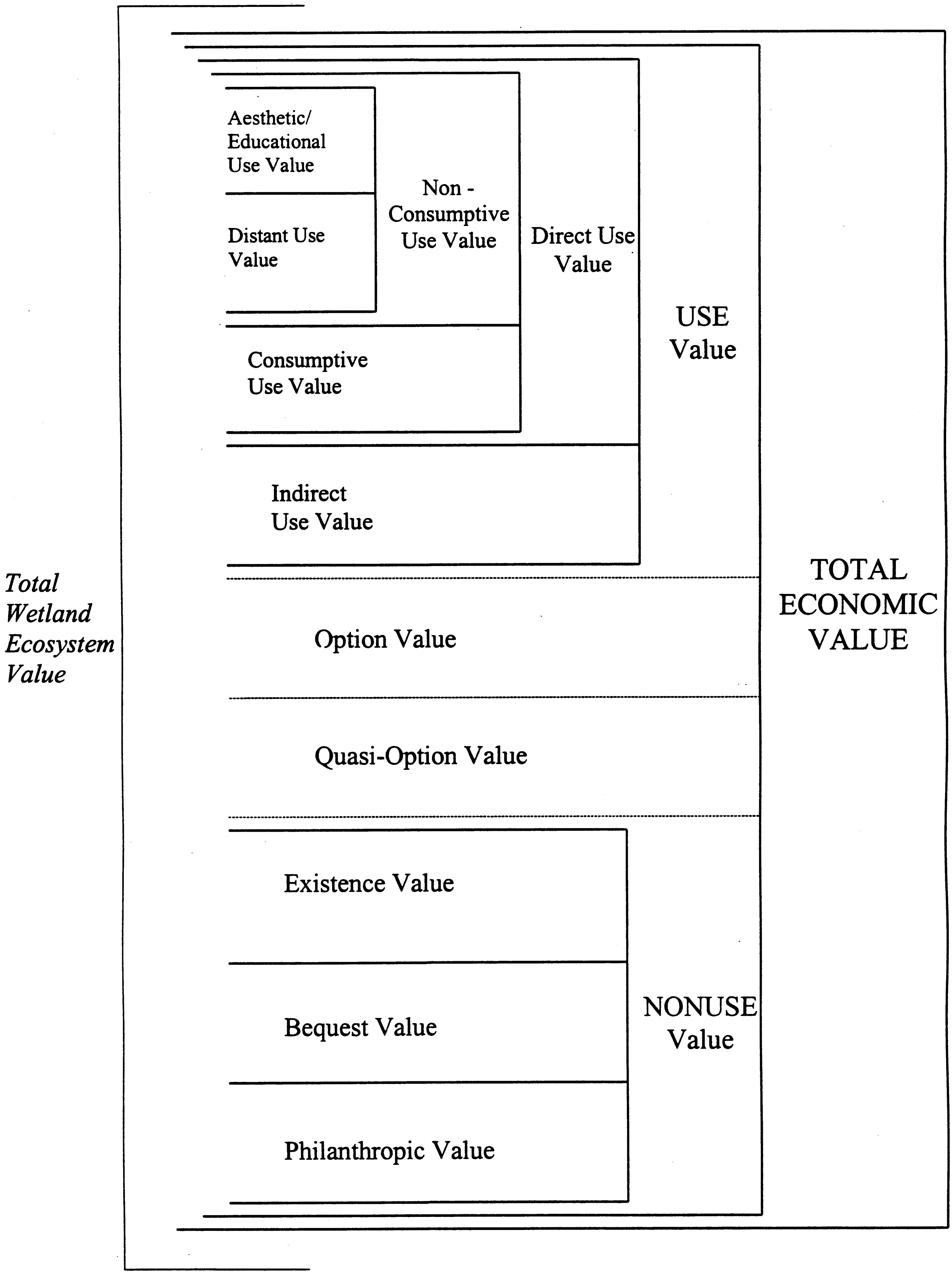
The danger of unpredictable and irreversible welfare losses can be more thoroughly qualified. Norton and Ulanowicz (1992) advocate a hierarchical approach to natural systems (which assumes that smaller subsystems change according to a faster dynamic than do larger encompassing systems) as a way of conceptualising problems of scale in determining biodiversity policy. For small scale disturbances there is thus greater possibility for substitutions within the ecosphere, and less possibility for large scale interferences. It is basically the combination of complementary relationships and gaps in knowledge that allows the deduction of a specific value for the stability of ecosystems. The preservation of the resilience of ecosystems protects man from incalculable welfare losses. Furthermore if the resistance and resilience of ecosystems depends on functional diversity, then the economic importance of structural components diversity lies in its function to minimise the risks of exogenous shocks.

The parallels between man made assets and biodiversity can be viewed as an important starting point for decision making in environmental policy. There are man made assets used for direct individual use (houses), others for productive uses (productive assets) as well as security assets serving to secure the above assets (insurance and social security systems). In the same way biodiversity can be attributed a function as a security asset beyond its productive and individual uses. Any evaluation based only on these former two value components will be incomplete since the potentially most important services of biodiversity result from its ecological protection functions, and further, these value components depend entirely on the ecological functionality of ecosystems. It is thus argued that biodiversity protection should focus not only on global extinction of single species but also on the consequences of altering the mix of species.

The question remains how does one fill this valuation gap within the framework of economic valuation. One approach has been to consider the ignorance about possible damages by the use of 'adders' in cost benefit calculations. However, such additions are arbitrary and do not consider the nature of the 'complementary relationships' and 'ignorance' characteristics, which become more important with increasing geographical scale of infringements.

Safe minimum standards as a decision rule are thus becoming more acceptable. Under this rule damage to biodiversity is to be avoided, as long as the opportunity costs for the current generation are not unacceptably high. Such an approach guarantees that the costs of biodiversity conservation are made explicit and a social decision concerning the limits of acceptable cost figures would have to be .

Figure 3.2 Components of economic value: wetland example



3.3 Biophysical values (indicators) and underlying scientific understanding of ecosystems

3.3.1 Introduction

This section looks at developing an integrated framework for biophysical ecosystem indicators in order to monitor and evaluate the impact of human activities on ecosystems. The work is based on existing scientific models underlying the compilation of wetland indicators. The concept of 'ecosystem integrity' is used as the organising principle behind the indicator framework. The break down in structural, compositional and functional system components, originally proposed in the context of biodiversity assessment techniques, is used to organise the indicator sets which cover three highly interrelated domains or aspects of ecosystems: landscape, water regime and biodiversity. The adopted holistic approach focuses on the interdependency and compatibility within and between indicator sets across different scales. The intention behind the development of the indicators is not for use as a backup method to evaluate the value of ecosystem functions and services, given situations where monetary valuation is difficult. Rather it is to monitor the health of the overall system and to provide quantitative information on the likely loss of functioning and functioning value.

One of the major problems when using indicators is the scientific uncertainty over whether they actually measure features of the environment that are of interest and that they change in some ecologically meaningful way with respect to environmental change (Norris and Norris, 1995). Ideally, one indicator can be constructed that captures the whole spectrum of relevant ecosystem attributes at different levels of organisation for sustainable management of that specific ecosystem, but this will almost never be the case. In fact, Landres *et al.* (1988) point out the danger of wildlife habitat management policy relying on a single indicator only (for example species indicators). Given the complexity of natural systems, the probability is small, even with adequate research, that a single indicator could serve as an index of the structure and functioning of an entire ecosystem.

Although agencies seem to prefer to promulgate and enforce regulations based on quantitative criteria, qualitative descriptions of qualitative changes in community structure are often the best indicators of ecological disruption (Noss, 1990). Here, we advocate numerical quantification of environmental changes as much as practically possible in terms of stress and effect indicators, but at the same time rely on qualitative descriptions of the intermediate changes or transitions between ecosystem states and ecosystem functions instead of assuming linear relationships.

The effect of pressures exerted by human activities on ecosystems can be measured by defining the relevant indicator spheres for ecosystem structure, composition and function. The sustainability of natural resource management or the impacts of environmental pressures on ecosystem integrity can subsequently be assessed in the following two ways.

First of all where possible, numerical stress and/or effect indicators can be related to ecological, biogeochemical and hydrological benchmarks or sustainability rules derived from natural scientific literature to determine ecosystem sustainability. For example, emission loads in an ecosystem can be compared to the assimilative capacity of the specific ecosystem.

Secondly, ecosystem integrity can be assessed by investigating the impact of environmental pressures on structural, compositional and functional ecosystem changes and the impact of these changes on each other. This is likely to be a more qualitative descriptive analysis. For example, human induced stress may alter ecosystem structure or composition. The impact of this change in ecosystem structure or composition on general ecosystem functioning has to be investigated to assess ecosystem integrity.

In the next section the specific indicator spheres of (wetland) ecosystem integrity will be presented.

3.3.2 Indicators of ecosystem integrity

Although a lot of research on different aspects of ecosystems exists, there does not seem to be much work which tries to bring these often separate pieces of investigation together under the same umbrella. Based on previous work, Angermeier and Karr (1993) provide a conceptual organisation of general ecosystem integrity with five classes of interacting factors: physiochemical conditions (e.g. salinity, nutrients), trophic base (e.g. system productivity, energy content of food), habitat structure (e.g. spatial complexity, vegetation form), temporal variation (e.g. seasonality, flow regime) and biotic interactions (e.g. predation, parasitism). This provides a comprehensive and seemingly complete framework to derive integrity indicators. Some of these factors refer to ecosystem structure and composition, while others refer to ecosystem processes.

Given the unique variables between water, land and biodiversity in most ecosystems, these three environmental media are used as the main organising features for the indicator sets. Indicator sets will be compiled for each of these features following the conceptual breakdown of ecosystem integrity into structure, composition and functioning (see Table 3.3). It is important to note the interdependency within and between ecosystem attributes (structure, composition and functioning) and ecosystem features (landscape, water regime and biodiversity).

In principle, each of these features and attributes can be monitored at their own specific but highly interrelated temporal and spatial scales for different levels of organisation. Ecosystem integrity can only be assessed if and only if these 3 indicator spheres are considered together, see annexes A and B.

Table 3.3: Ecosystem integrity indicator spheres (wetland example)[1]

Ecosystem feature	Ecosystem attribute	Indicator sphere
	Structure	wetland area structure
Landscape	Composition	land form and cover
	Functioning	land use changes
	Structure	Hydrology
Water regime	Composition	biogeochemical water properties
	Functioning	water constituents flux
	Structure	food web trophic structure
Biodiversity	Composition	Keystone species and umbrella species
	Functioning	energy transfer between trophic levels

[1] Annex A discusses the determination of indicators of ecosystem integrity in more detail. Annex B discusses the research work on derivation of indicator for wetlands

4. Socio-economic values (indicators): the role and limitations of economic valuation and implications for decision making systems

4.1 Introduction

Section 1.3 introduced the conceptual link between ecosystem functions and their importance to society, and considered the general implications of this ecological perspective for economic approaches. Section 3.3 considered the definition of biophysical indicators to use alongside economic valuation work. This section returns to the discussion of socio-economic evaluation, to consider the issues in more detail.

Biophysical quantification of ecosystem sustainability indicators will not in itself be a sufficient evaluation from a dynamic systems perspective. Socio-economic indicators are also needed. The essence of an overall socio-economic evaluation is to determine how society is affected by the functions an ecosystem might perform - the function itself is not intrinsically valuable. It will therefore be necessary to assess features of anthropogenic regimes and how these respond to changes in ecosystem functioning. We look at the role and limitations of economic valuation, its alternatives, and implications for decision making systems.

In environmental economics, an individual preference-based value system operates in which the benefits of environmental gain (or the damages from environmental loss) are measured by social opportunity cost (i.e. cost of forgone options) or total economic value. The assumption is that the functioning of ecosystems provides society with a vast number of environmental goods and services which are of instrumental value to the extent that some individual is willing to pay for the satisfaction of a preference. It is taken as axiomatic that individuals almost always make choices (express their preferences), subject to an income budget constraint, which benefit (directly or indirectly) themselves or enhance their welfare. Households are assumed to maximise well-being deriving from different sources of value subject to an income constraint. Their private willingness-to-pay (their valuation) is a function of prices, income and household tasks (including environmental attitudes) together with conditioning variables such as household size. The social value of environmental resource committed to some use is then defined as the aggregation of private values. Nature conservation benefits should be valued and compared with the relevant costs. Conservation measures should only be adopted if it can be demonstrated that they generate net economic benefits.

Other environmental analysts, on the other hand, either claim that nature has non-anthropocentric intrinsic value and non-human species possess moral interests or rights, or that while all values are anthropocentric and usually (but not always) instrumental the economic approach to valuation is only a partial approach. 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. Nevertheless, for some people it is feasible and desirable to manage the environment without prices. According to O'Neil (1997), for example, conflicts of values in forestry and biodiversity management issues in the UK are resolved through pragmatic methods of argument between botanists, ornithologists, zoologists, landscape managers, members of a local community, farmers etc.

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

As citizens, individuals are influenced by held values, attitudes, and beliefs about public-type goods and their provision. In this context, property rights (actual and/or perceived), social choices and moral concerns can all be involved in a nature conservation versus development conflict. The polar opposite view to the conventional economic approach would hold that the very treatment of ecological assets such as biodiversity in terms of commercial norms itself is part of the environmental crisis. The argument becomes one of the 'proper' extent of market influences and commodification (O'Neil, 1997). Market boundaries should not from this perspective be extended to cover as many environmental assets as is possible. Instead society should give greater consideration to the nature of deliberative institutions for resolving environmental problems and of the social and economic framework that will sustain them (O'Neil, 1997). The counterbalancing argument would be that some environmental goods/services which have a mixed public/private good set of characteristics (e.g. forests watersheds, 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). Figure 4.1 summarises three highly simplified and probably overlapping worldviews about the valuation and assessment of environmental quality.

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 then with the situation as it would be without the option. The difference is the incremental net benefit arising from the project investment. Cost Benefit Analysis 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. environmental ones, are translated into monetary terms. As a rule, a project is efficient if total benefits exceed total costs.

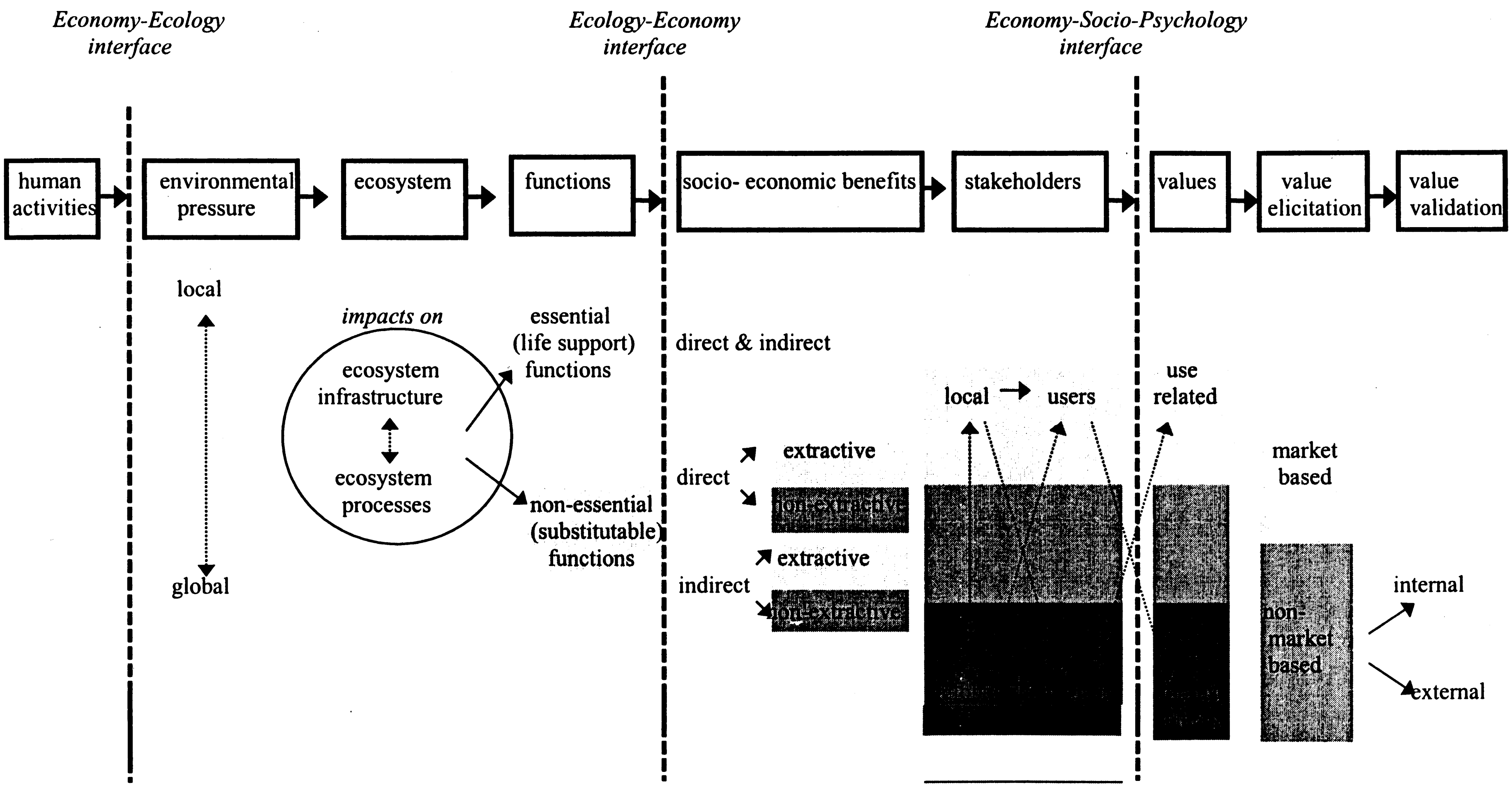
Diagram 4.1 is an attempt to depict the general framework of the environmental valuation problem and will be used to increase the transparency and hence the legitimacy in the monetary estimation of certain environmental values and their use in transfer exercises.

The figure shows that human activities exert pressures on the environment resulting in specific changes in ecosystem structures and functioning. These environmental changes in turn have an impact on human welfare. Environmental valuation practitioners have only in a limited way addressed the question which of the environmental impacts of human activities on human welfare can and cannot be translated in monetary values in a meaningful and hence reliable way.

The shaded areas in indicate the domains where monetary environmental valuation might play a role in CBA. The darker shades indicate that there is a significant increase in difficulty in applying the above mentioned valuation techniques and methods in these areas. Between local and global beneficiaries, users and non-users, and use related and non-use related values there is a large undefined grey area where the application of environmental valuation techniques may be more or less appropriate. In the diagram, the areas have been depicted as equally sized, but in the end it may very well appear to be that the darker shaded areas are larger than the less dark shaded areas.

A range of valuation techniques exists for assessing the economic value of the functions performed by ecosystems, and these are detailed in Table 4.1. Many ecological 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 non-market valuation techniques. More detailed information on the underlying theory and practical implementation of these techniques can be found in a number of general texts including Braden & Kolstad (1991), Bromley (1995), Dixon & Hufschmidt (1986), Freeman (1993), Hanley & Spash (1993), Pearce *et al.* (1994), Randall (1987), Turner (1993b), and Turner & Adger (1996).

Diagram 4.1: A simple general framework for monetary environmental valuation including environmental value transfer



Key to symbols: see text.

Table 4.1 Valuation Methodologies Relating to Ecosystem 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)	Ecosystem treated as one input into the production of other goods: based on ecological 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 Ecosystem function were not present; eg 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 ecosystem to its original state. A total value approach; important ecological, temporal and cultural dimensions	✓	✓	✓[3]

Notes:

[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 wetland or the time frame for restoration.

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 costs analysis.

4.1.1 Limitations of economic valuation

Ecosystem perspective

The links between ecosystem functions and values was shown in figure 3.1 Under such a functional approach to economic valuation it is clear that ecosystem uses, or the output of physical products or services, form the essential link between ecosystem ecology or functioning and ecosystem values. Non-use values will be independent of use, although they will be dependent upon the essential structure of the ecosystem and functions it performs. Whatever the typology adopted to describe types of economic value, it will always be contingent on the ecosystem performing functions that are somehow perceived as valuable by society. Functions in themselves are therefore not necessarily of economic value; such value derives from the existence of a demand for these functions or for the goods and services they provide. It is therefore important to identify how particular functions might be of use, rather than simply the degree to which the function is being performed. The extent of demand for the products or services provided, or the effective 'market', needs to be assessed if the full extent of economic value is to be assessed.

Given the general typology for the assessment of ecosystem benefits provided in Figure 3.1 the first step is to compile a complete list of all the relevant boundary conditions for the ecosystem. These are those characteristic properties that describe the area in the simplest and most objective possible terms. Examples of characteristics include the biological, chemical and physical features that would describe ecosystem such as size, shape, depth, climate, species present, vegetation structure and the natural processes occurring there. Characteristics, singly or in combination, give rise to benefits, which may be potential rather than currently realised.

From an anthropocentric viewpoint all ecosystems can be classified in terms of their structural and functional aspects (Westman, 1985; Turner, 1988). Ecosystem structure is defined as the tangible items such as plants, animals, soil, air and water of which it is composed. Thus structural benefits (of instrumental value to humans) include fish, waterfowl, peat, timber, reed and fur harvests as well as non-consumptive use benefits such as recreation and research or education. By contrast, ecosystem processes are encompassed by the dynamics of exchange of means of energy. The processes are subsequently responsible for the services - life support services, such as assimilation of pollutants, cycling of nutrients and maintenance of the balance of gases in the air.

The task of evaluating the structure and functioning of an ecosystem implies that we know fully what the ecosystem does and what that worth is to us. The worth of ecosystem structure is generally more easily appreciated than that of ecosystem functioning. To evaluate functions such as nitrogen fixation, nutrient and soil retention, gas exchange, radiation balance, pollution absorption, and other ecological processes for any given segment of landscape, pushes present ecological knowledge beyond its bounds. Even ecosystem structure is incompletely known. To evaluate the worth of the insect fauna, or soil fungi, when many of these species have never even been described taxonomically, taxes human knowledge beyond current limits (Westman, 1985). The preservation of ecosystem processes is as important a goal for conservation as is the preservation of ecosystem structure. The science of ecology has now elucidated ecosystem processes to the extent that some management principles are evident, yet much research on ecosystem structure and functioning is still needed.

Ecosystem structure provides humans with goods or products which involves some direct utilisation of one or more characteristics of a wetland or other habitat. The processes in an ecosystem, on the other hand, provide ecologically related services to humans, so that some aspect of an ecosystem supports or protects a human activity or human property without being used directly. One 'rule of thumb' for recognising these services is that they provide a benefit that people gain without necessarily having to go to the ecosystem.

The significant question here is: "can all the benefits from all the classes of ecosystems be classified as goods, products and services?" It is evident that there are strong linkages between the types of benefits. For example, the sound functioning of an ecosystem through efficient nutrient, sediment and toxicant removal is necessary to ensure viable fish production. Nevertheless each of these benefits provides a distinct positive value to the overall system, although the need to ensure against double counting cannot be overstated. An assessment of the complete range of benefits at a site using a standard classification of benefits is an essential step before the overall value of an ecosystem can be derived. It is important to remember, however, that the social value of an ecosystem (total system value), may not be equivalent to the aggregate private total economic value of that same system's components (see section 3.2.3).

Irreversibility, thresholds and the need for safe minimum standards

An important aspect of the economics-science interface is the existence of thresholds and the potential for irreversible change. Where the additional change in a parameter has a disproportionate effect, this might be associated with relatively high economic values. And if the change is irreversible, account needs to be taken of the uncertain future losses that might be associated with this change, and the possible imposition of a Safe Minimum Standards (SMS) decision rule (Ciriacy-Wantrup, 1952; Bishop, 1978; Crowards, 1996). This recommends that when an impact on the environment threatens to breach an irreversible threshold, that the conservation option be adopted unless the costs of forgoing the development are regarded as 'unacceptable'. It is based on a principle of minimising the maximum possible loss, rather than cost-benefit and risk analysis which is based on maximising expected gains. Given our absolute uncertainty regarding the benefits in the future that might have been derived from the threatened resource, our maximum possible loss must be associated with loss of that resource. We can calculate the net benefits that we expect to derive from the development project which threatens this resource, and so long as forgoing these benefits is regarded as an acceptable sacrifice, the conservation option is always preferred when faced with potentially irreversible damages.

Clearly the critical factor in SMS is what is regarded as an 'unacceptable' sacrifice of present benefits for the sake of possible future losses. The degree of sacrifice entailed involves a full cost-benefit assessment of the development option, including the estimable costs of damage to the environment. It is then a broadly political decision within the constraints of society's other goals, as to whether avoiding potentially massive, but wholly uncertain, future costs can be justified. In this sense it provides a mechanism for incorporating the Precautionary Principle into decision-making, where 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 a).

The concept of Safe Minimum Standards has usually been applied to endangered species. In this manner it may well be applicable to a number of ecosystems given their role in supporting a variety of threatened species. However, it could equally well apply to irreversible impacts threatening ecosystems as a whole. One complication is to identify what is a truly irreversible change in the ecosystem, since any change that can be reversed in the future will not necessarily entail the maximum possible costs. It will also be necessary to determine whether or not thresholds in current ecosystem functioning exist, and whether these may be threatened by proposed developments. Where it is discerned that thresholds of ecosystem functioning are threatened with irreversible change, SMS as a decision framework that gives more weight to concerns of future generations and promotes a more sustainable approach to current development, might represent an appropriate supplement to purely monetary analysis.

Other/ practical issues

Scale

It is important to determine initially what the scale of assessment is going to be. This may involve impact analysis of a limited number of affected variables associated with an isolated external impact. Where more general changes such as alternative uses of the ecosystem are being considered, partial analysis of a number of integrated parameters may be required. A total valuation will generally entail considerable effort but may be appropriate where, a) the ecosystem as a whole is threatened, and, b) the benefits deriving from an alternative use outweigh the benefits estimated from only a partial estimation of ecosystem value.

The geographical scale of assessment will be important. The relevant population for an economic assessment will depend in part on the type of function that is being valued. Direct use values will generally involve some contact with the ecosystem itself, although individuals may travel considerable distances in order to make use of the ecosystem. Indirect use values may be site-specific in terms of those who benefit, non-use values are likely to be derived over a wide geographical range, but are likely to be subject to 'distance decay' away from the site.

Temporal scale in combination with the rate of discount applied will influence the present value of benefits attributed to ecosystem functions. Calculating expected future costs and benefits involves estimating future demand for the ecosystem's functions. This will necessarily be unknown but assessing likely scenarios and applying sensitivity analysis can provide a range of possible values.

Aggregation and double counting

If each output provided by an ecosystem is identified separately, and then attributed to underlying functions, there is the likelihood that benefits will be double counted. Benefits might therefore have to be explicitly allocated between functions. For instance, Barbier (1994) notes 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 groundwater recharge, so that combining these values would overestimate the feasible benefits to be derived from the ecosystem. In the case of reed bed management, conservation goals may require alteration of harvesting practices that reduce gross margins, possibly even to the extent that margins become negative. Clearly, combining the potential benefits from harvesting and from biodiversity conservation without considering the links between the two can overstate the benefits. It may be possible that some functions, are complements rather than competitors. For instance, nutrient retention could promote biomass production and the possibilities for harvesting, thereby adding to the value of the nutrient retention function.

4.1.2 Combining expert (professional) scientific understanding, non-expert (non-professional) environmental knowledge and environmental values

A combination of quantitative and qualitative research methods is advocated in order to generate an optimum blend of different types of policy relevant information. This applies to both the biophysical assessment of management options, and especially for the socio-economic values people hold associated with the environmental changes these management options entail.

A combination of quantitative and qualitative approaches is expected to produce the most useful and meaningful information when compiling indicators. Indicators are commonly understood as quantifiable variables which provide information about changes in for example environmental conditions. The variable itself may describe an environmental state at a certain point in time and an analysis of these variables over time will provide information about the relevant changes and the rate of change. However, indicators can also be qualitative and descriptive in nature.

Although agencies seem to prefer to promulgate and enforce regulations based on quantitative criteria, qualitative descriptions of qualitative changes in for example community structure are often the best indicators of ecological disruption. In practice it is only possible to numerically quantify some environmental changes while at the same time complementing these indicators with qualitative descriptions of the intermediate changes or transitions between ecosystem states and ecosystem functioning. (Brouwer et al. 1998). The same caveat applies to the construction of relevant social and economic indicators.

Social research dependent on quantitative research methods and techniques is premised on the assumption that opinions, feelings, perceptions, beliefs, attitudes or behaviour can be expressed in meaningful numerical ways within a given context. It is most often criticised for its overly reductionist character in the face of real world complexity and diversity, *i.e.* social, cultural, economic, political and environmental. Its technical nature may also act as a shroud, obscuring its 'proper' interpretation by the public.

Qualitative research methods, on the other hand, are, in principle, more comprehensive in their coverage of the variety of contexts found in society. But such research usually produces a vast amount of ethnographic data not amenable to scrutiny via traditional statistical or related analysis as in quantitative approaches. Consequently, interpretation of the results is perhaps an even more difficult task and the risk of manipulation and value judgement masking no less apparent.

More research is needed since the individual and group based approaches place the whole process of eliciting environmental values, monetised or not, in different social settings and therefore provide us with different kinds of information. Typically, qualitative research will provide in-depth information on fewer cases, whereas quantitative procedures will allow for more breadth of information across a larger number of cases. A combination of both approaches offers future promise for environmental valuation.

The use of either or both approaches depends on the type of information policy and decision makers are looking for in specific policy domains, but also the type of information the public is able to deliver and how much the public is willing to participate in public consultation. In this respect, it is furthermore important to distinguish clearly between social and social scientist preferences for different approaches of public consultation.

5. Case study of the evaluation of wetland functions on the Norfolk Broads

The research project involved an attempt to apply the concepts reported in earlier sections to a specific case study area. The Norfolk Broads Wetlands has been the subject of several ecological and socio-economic studies. The Norfolk Broads perform a variety of functions valued by a variety of stakeholder groups, living and working in the area or visiting the area. The Broads wetlands provide a buffer against extreme hydrological conditions, providing water storage in times of flood and water release during drought. Wetlands also have the capacity to change water quality through the removal of chemical pollutants such as nitrogen and phosphate. A third major function is the provision of a nationally and internationally important wildlife habitat.

The main wetland functions are presented in Table 5.1. These are categorised according to whether they are hydrological, biogeochemical or ecological functions. The table details the biophysical structure and processes maintaining the functions, their socio-economic uses and benefits and threats to future availability of the functions.

This section does not provide detail on economic valuation techniques or their applications, though the research project did look at several case studies in the Broads which are summarised in section 5.4, see references for further details. Instead, it focuses on the considerations and thought processes that are necessary in making the link between ecological functions and their significance to society. It concentrates on the identification of the issues one is trying to measure, rather than the method of measurement.

Table5.1: Wetland functions and associated socio-economic benefits in the Broads

Habitat	→ Biophysical Structure or Process Maintaining Function	→ Function	→ Socio-Economic Use and Benefits	Threats
		Hydrological Functions		
	short and long term storage of overbank flood water and detention of surface water runoff from surrounding slopes	flood water retention	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
Rivers lakes marshes fens	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		
rivers lakes	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
Fens	in situ retention of C	peat accumulation	Fuel, Paleo-environmental data source	overexploitation, drainage
		Ecological Functions		
Rivers, lakes marshes fens woodland	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
Marshes Fens	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).

5.1 Socio-Economic Valuation of Hydrological Functions[1]

5.1.1 Flood Water Control

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

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

2. 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).

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

4. 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 expanse 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. The choice of method is likely to depend, in part, on time, resources and data that available for the investigation.

The potential for downstream flooding:

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 (eg buildings or farming) 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.

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[1] This section draws heavily on Crowards & Turner (1996)

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

Assessing the wetland's influence on downstream flooding:

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 and hence reduction in river volume 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 significant downstream peak discharge reduction 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 watershed-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 & Smith (1984, p.5).

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

e.g. historical episodes; river authority 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; river authority predictions or records; 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 floods (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 or intensive agricultural land use is under threat, damage costs could be considerable and longer term.

Velocity reduction and erosion control

By diverting flood waters for possible future, more gradual, release, a river marginal wetland may serve not only to reduce flooding damage, but also to prevent erosion caused by peak river discharges. Furthermore, some wetlands may serve to reduce the velocity of flow not only during high discharge episodes but on a more continual basis, thereby limiting erosion downstream.

Such additional or external effects of substitute/shadow projects will influence estimates of project costs and thereby the value associated with maintaining the flood control function. If only the financial costs are incorporated (which are relatively easy to assess and are therefore more generally available), then the implicit assumption is that these options involve no environmental (or other) costs or benefits other than the provision of flood protection.

5.1.2 Groundwater recharge

A wetland's ability to recharge groundwater will only be of value if the groundwater is then of some benefit to society. This may be in terms of direct abstraction of the water for irrigation or domestic use, or may be more indirectly useful as, for instance, in maintaining water table levels and preventing salinization. Whatever the subsequent use of the groundwater may be, it will be important to identify a use, or perhaps non-use motivations for maintaining supplies, if economic value is to be associated with the wetland's recharge function. Non-use values could be attributed by the current generation to maintaining groundwater supplies for the sake of subsequent generations, but otherwise, simply recharging groundwater that might one day prove to be of use will not be associated with any economic value unless this use can be anticipated in the foreseeable future.

Furthermore, once benefits to be derived from groundwater recharge by the wetland have been identified - which may itself be an extremely complex task involving analysis of subterranean hydrological systems - the **degree** to which the wetland's recharge contributes to the maintenance of groundwater levels will need to be assessed. This is a problem common to many off-site benefits associated with wetland functions, where the value of these off-site benefits needs to be apportioned among contributing sources, including the wetland. The degree of wetland contribution to the benefits derived from groundwater will depend, among other things, on whether there is any apparent shortage of groundwater and if it is being abstracted faster than it is being regenerated; how much the wetland contributes to overall supplies; and whether the wetland's contribution is of direct benefit in maintaining the quantity and quality of specific local supplies. As a consequence of the complexities involved, it may not be possible to assess the economic value of the groundwater recharge function as part of a standard evaluation of a wetland, unless considerable previous scientific investigation has been carried out into the underlying groundwater system.

The techniques involved in assessing the economic value of groundwater recharge will be much the same as those outlined in the subsequent section on 'Surface water generation'.

5.1.3 Groundwater discharge

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 can therefore be valued with the same techniques those applied to surface water. 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. Characteristics of discharged groundwater, such as temperature and chemical constituents, might influence other wetland functions such as primary productivity and the ability to retain nutrients, but any benefits that may be associated with these will be included in valuation of the 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 watershed level. In this case, benefits associated with surface water could be linked to its previous source, thereby attributing value, perhaps, to ecosystems that have facilitated the previous recharge of groundwater elsewhere.

5.1.4 Surface water generation

The surface water generation function of a wetland may be of value directly for on-site use or in providing off-site benefits via maintenance of downstream water flow. A number of the on-site benefits that derive from wetlands having available surface water - a necessary requirement for any 'wetland' at least for part of the time - are considered in other sections. For instance, the reliance of characteristic wetland ecology on surface water and anaerobic conditions resulting from inundation, with subsequent ability to retain excess nutrients, are considered in the sections on 'Ecological functions' and 'Nutrient retention', respectively. The only on-site value considered here is that of surface water abstraction from the wetland, a form of consumptive use value. The off-site benefits of a wetland's ability to increase surface water are more varied, to the extent that these benefits are not considered in the valuation of any other function of the wetland, and might include maintaining habitats, providing aesthetic and recreational value, and direct abstraction.

Downstream habitat and biodiversity maintenance are considered below only 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 in the section 'Ecological functions'. Direct abstraction can be valued in a similar fashion whether it is downstream or within the wetland, although as with all off-site benefits, the degree to which the wetland in question contributes to the amount of water available (and so, for instance, to biodiversity maintenance, recreational enjoyment and the potential for abstraction) is crucial to attributing value to the wetland. The downstream benefits of wetlands surface water generation are generally associated with maintaining a steady flow of water that alleviates often seasonal extremes of low river flow. This derives from wetlands acting as a 'sponge', holding water when flows are high, and releasing it gradually over a longer time period. However, it is not clear that wetlands do necessarily act in this fashion (Bardecki, 1984), and nor do there appear to be any studies that have sought to attribute a value to wetlands in this regard. Techniques are outlined here that could be used to estimate the value of maintaining off-site water supplies if it can be shown that the wetland is indeed performing this function and the degree to which these off-site supplies are dependent on the wetland acting in this capacity.

'Off-site' here is taken to mean both downstream - ie a watercourse supplied at least partially by the wetland - and groundwater sources, since although groundwater recharge is a separate wetland function, the valuation techniques are essentially the same for surface water and groundwater abstraction.

An important aspect of benefits deriving from on-site abstraction of surface water is the degree to which the wetland is able to support this activity without suffering damage to the ecosystem. Clearly there will be minimum requirements for the level of water remaining in the wetland if it is to retain its essential ecosystem characteristics and continue to perform a range of other functions. The abstraction of water beyond threshold levels may have highly uncertain but possibly irreversible consequences, suggesting that concepts of safe minimum standards and sustainability constraints might be usefully introduced. Certainly, any economic valuation should take into account the sustainability of any such consumptive use of wetland resources and the likely time frame over which these resources (and possibly, as a result, other wetland functions) might be exhausted

5.2 Economic valuation of biogeochemical functions

5.2.1 Nutrient retention

Assessing the economic value of the nutrient retention function of a wetland involves:

1. determining the degree to which nutrient release into water sources is reduced by the wetland;

2. identifying where any adverse effects that increased nutrient levels might have will occur, and what these adverse effects are likely to be;

3. linking possible (quantified) increases in nutrient release to nutrient levels at sites where adverse effects are expected;

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

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

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

When the extent of an impact is known (or approximated), for instance in terms of increased eutrophication in water bodies, levels of pollutants in drinking water, or deterioration in species populations, the economic consequences of damage to the wetland that induces such an impact can be assessed.

While the valuation techniques involved will be the same, the distinction between nutrient **retention** and nutrient **removal** is important. Since retention implies a comparatively short time horizon, if no subsequent **export** 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 waterstream, must be considered. Presumably, if no export is occurring then continual nutrient retention cannot be sustained indefinitely. This suggests that issues such as sustainability and safe minimum standards need to be incorporated into the economic evaluation process. These concepts are considered in detail in sections above.

5.2.2 Nutrient export

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. 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. Therefore, while the initial valuation techniques will be the same for both nutrient retention and export (hence, the previous section 'Nutrient retention' should be referred to for valuing the nutrient export function), the level of nutrients that can be diverted from the water stream by the wetland may alter according to which of these two functions is being performed. 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. These concepts are considered in detail in previous sections.

A further consideration in the context of the nutrient export function is the manner in which export occurs and where the nutrients end up having been exported. So, for instance, if nutrients are removed via physical export downstream, might this 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.

5.2.3 Sediment retention

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 reducing the load of sediment in waters downstream. Other potential benefits **within** the wetland, such as improved water quality and increased biological productivity, will be included in the valuation of functions such as biodiversity maintenance and biomass harvesting . Valuation of the damages resulting from sediment loading (or, from another possible viewpoint, benefits of reduced loading due to the wetland's sediment retention function) is considered in turn:

Off-stream damages

1. The ability of a wetland to mitigate downstream flooding, including 'sediment-related flood damages', was considered in the section on 'Flood water control'. A number of valuation techniques are analysed, which will be equally applicable to the effect of reducing flood damages as a result of limiting sediment loads. Difficulty may arise in distinguishing between damage attributable to sediment (including the 'indirect' effect that it may have in increasing the height and volume of floods) from flood damages in general. This may be even more pronounced with valuation techniques that are based on **benefits** of reduced flooding, such as hedonic property pricing and contingent valuation, rather than on the costs of flooding (i.e. damage costs avoided). However, given that the overall purpose of the valuation exercise is to place a value on the wetland as a whole, there may be little advantage in assessing the specific contribution of reduced sediment load to the benefits of flood mitigation.

2. The benefits that reduced sediment loads might have in terms of mitigating damages to 'water conveyance facilities', 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 likely to be the most appropriate valuation technique.

3. Increased costs associated with 'water treatment for municipal and industrial use' resulting from increased sedimentation, in terms of providing additional treatment facilities to substitute for potential loss of the wetland sediment retention function, could be used to assess the value of this function. Such a substitute cost approach is outlined in the section 'Nutrient removal' with specific reference to water treatment.

4. 'Other off-stream effects' might be most readily estimated in terms of additional costs incurred by industrial and municipal users for water that is not fully treated, and in terms of productivity losses for irrigated agriculture where fine silt particles in irrigation water can serve to seal the soil surface. Possible benefits that additional sediment in the water might provide, such as increasing its cooling capacity for energy generators (thereby reducing costs) and increasing fertility in agriculture (thereby improving productivity), should also be taken into account.

The majority of the impacts that increased sediment loads might have are therefore similar to those considered in other sections. The valuation techniques that can be applied to the loss of a wetland's sediment retention function are subsequently covered in other sections, as indicated.

5.2.4 Peat accumulation

The main benefit of a wetland's ability to accumulate peat may be in terms of providing a long-term store for carbon. 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.*, 1992). While accumulating peat might be beneficial in other ways, such as creating a unique form of habitat or providing a source of fuel or horticultural materials, these are not considered in this section. Some of these other benefits, such as providing habitat, will be covered in other sections of the valuation procedures (in this case, section 3.1.c, 'Biodiversity maintenance'). Others, such as in creating a source of fuel, are not relevant benefits in the time scale involved, and in any case extraction of peat represents a wholly unsustainable degradation of the ecosystem. The considerable length of time that it takes peat to accumulate rules out any benefit being attributed to the accumulation function based on extraction, within the temporal framework of an economic study.

The value of peat accumulation as a store of carbon is not easily assessed. The degree of accumulation and of storage need to be estimated and their significance in terms of global carbon emissions assessed. For many wetlands, the rate of carbon **accumulation** (probably less than one tonne per hectare per annum) will not result in significant economic benefits. However, peat wetlands acting as a **store** of previously accumulated peat may be associated with considerable value in terms of containing a potential source of atmospheric carbon. Were a peatland to be converted to another use, there may be a considerable release of carbon that would represent an external cost to the conversion project, in addition to any loss of the range functions performed by the ecosystem. Therefore, while the slow rates of current accumulation might not generate significant value, the continued storage of carbon could be associated with considerable economic benefits. 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.

5.2.5 Salt accumulation

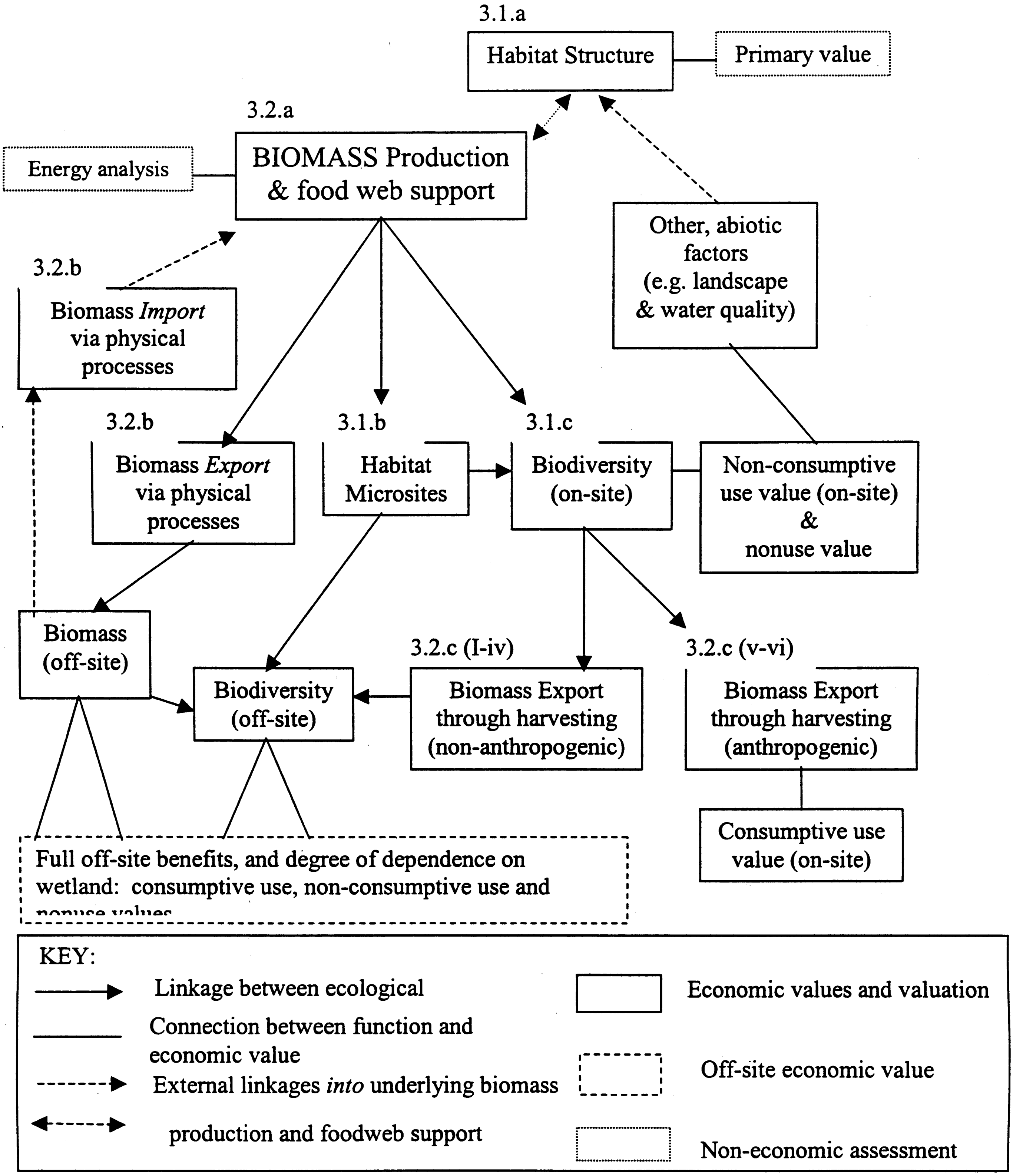
The accumulation of salts by wetlands is not of any apparent benefit as a function of the wetland. However, high concentrations of salt in the ecosystem might suggest that conversion of the wetland to agriculture might be highly unsustainable. High levels of salt will inhibit non-wetland plant growth thereby limiting the potential of this land for any form of dry-land agriculture. The opportunity costs of maintaining the wetland, therefore, in terms of an alternative use involving agriculture, are likely to be low for wetlands with significant salt accumulation.

5.3 Economic 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. But note the discussion of TEV and Total Environmental System Value in section 3.2.3. An example of the relationship between individual wetland ecological functions and economic valuation is illustrated in Figure 5.2, and is outlined below. Essentially, there is considerable overlap in the list of 'Ecological Functions' - so, for instance, biomass export (whether via abiotic or biotic media) is necessarily dependent upon biomass production (allowing for any biomass import) - and, 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, then, estimation focuses specifically upon biodiversity maintenance and anthropogenic harvesting of biomass.

The schematic of Figure 5.2 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 (see discussion on glue values – section 3.2.3). 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 result of such export.

Figure 5.2 Ecological Functions and Economic Value



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.

5.3.1 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. It is also possible that value might be associated with habitat or ecosystem structure; so-called primary value (see section 3.2.3).

5.3.2 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 current 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 & 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 of preservation being highlighted as maintaining a site for gull nesting.

5.3.3 Maintaining biodiversity

It is 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, (anthropogenic) export of this biodiversity, 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 biomass production and foodweb support and be dependent upon overall ecosystem health and habitat structure, these functions 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.

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.

Identifying potential value in terms of biodiversity maintenance

Activities that result in the deliberate export or removal of material are dealt with in section 5.3.6, '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.

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 recreationists 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.

Non-economic criteria

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

5.3.4 Biomass production

Biomass production is the basic requirement for 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 & 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 the wetland 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.

5.3.5 Biomass import and export via physical processes

The transfer of biomass out of the wetland to another ecosystem is not in itself a valuable function. However, if the biomass is of benefit to society in these alternative ecosystems, then this benefit may be attributed in part to the wetland. This will require estimating economic values derived from the alternative ecosystems and the degree to which these are reliant upon the export of biomass from the wetland. The approach will be similar to that outlined under 'Provision of microsites', although depending upon the relevant benefits, estimation techniques may derive from sections on 'Hydrological Functions' and 'Biogeochemical Functions', as well as the current section on 'Ecological Functions'. Given the considerable amount of work involved in estimating benefits in an entirely new ecosystem, quantifying the benefits of biomass export from the wetland is unlikely to be required unless the benefits are likely to be considerable.

The transfer of biomass **into** the system, or biomass import, is also not a valuable function in itself. Just as biomass export might result in functions off-site that are beneficial to society, biomass import may contribute to functions within the 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 watershed 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 watershed.

5.3.6 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 might be associated with benefits derived from off-site locations - i.e. 'biomass export via physical processes' and '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 - which is more likely to be associated with acting as a habitat microsite than with the natural export of biomass - will analysis be justified. Such analysis 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.

5.3.7 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. 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, shrimp 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 preservation, 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 & 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.

5.4 Summary of case study findings

At the wetland ecosystem level, it has proved possible to confirm a coherent set of links between wetland boundary conditions, structure, processes and functions, as well as the consequent outputs of goods and services from which humans derive both use and non-use value. The precise quantification and valuation of multiple wetland functions and outputs is not, however, straightforward. Function overlap (a double counting problem) and conflict (one use precluding another use) and the delineation of the overall ecological system integrity ('infrastructure' or 'primary' value of wetlands) present a number of complications. Nevertheless, the research has confirmed the importance of the 'functional' approach and the related policy objective of 'maintaining functional diversity' at the catchment/landscape scale in order to assess the value of nature conservation in the context of a wetland ecosystem.

At the landscape scale, the D-P-S-I-R auditing framework has also proved to be a very useful device. It has allowed the research to coherently address a range of issues resulting from regional/local environmental pressures and resource use conflicts.

From a methodological perspective it become clear that a mixed quantitative/qualitative approach is necessary in order to address the various stakeholder conflict situations and their possible mitigation such that the value of nature conservation can be found. The use of cost-benefit analysis (CBA) in environmental decision-making and the contingent valuation (CV) technique as input into CBA to elicit monetised environmental values has stimulated an extensive debate. Critics have questioned the appropriateness of both the method and the technique. Some alternative suggestions for the elicitation of environmental values are based on a social process of deliberation. However, just like traditional economic theory, these alternative approaches may be questioned in terms of their implicit value judgements. The view taken in this case study research was that instead of a priori assumptions, research efforts should be focused on the processes by which actual public attitudes and preferences towards the environment can best be elicited and fed into the policy process. The Broadland work found support for both the individual WTP based approach and a participatory social deliberation approach. This suggests that a combination of both approaches can be appropriately deployed. More work is, however, required in order to be able to state more confidently precisely which valuation contexts are most appropriate and what the exact configuration of the mixed methodology should be used.

This is confirmed in the case study, which covers work related to the complex coastal wetland, known as Broadland, in the south East of England. A suite of interdependent activities relate to the utilisation of the Norfolk and Suffolk Broadland both as a source of high quality recreation/amenity, biodiversity maintenance, flood alleviation and as a supply of reed/sedge biomass for an energy recovery scheme and other potential end uses.

The case study has considered a number of examples to illustrate the procedures involved in assessing the value of ecosystem functions, functioning and nature conservation. A sophisticated contingent valuation survey design was used in order to obtain monetary (willingness-to-pay) values from a large sample of recreational users. Sample respondents were asked how much they would be willing-to-pay (WTP) in extra taxes per year in order to ensure the preservation of Broadland from the effects of increased flooding. Results suggest that a majority of individuals are prepared to pay to protect the wetland (from saline flooding) and do not object to the monetary valuation process itself.

Further examples are related to the Broadland wetland context. These include the compilation of a number of environmental indicators for use in wetland management. The indicators will have to be based on existing and available date. Initial work has identified potential indicators for wetland structure (e.g. water level regime, level-area volume relationships and water balance); wetland composition (e.g. sediment load, organic load and nutrient load, toxicant load, oxygen level and bacteria, viruses and water-borne parasites, temperature); and wetland function (e.g. turnover rate).

A key to resolving present failures thus seems to be behavioural change at the local level. Increased scientific knowledge of wetland ecosystems and their benefits to society has to be gained hand-in-hand with efforts to increase public awareness of these benefits. Such a communication is however only likely to be successful if due account is taken of the potential difference in worldviews between the scientists and local people. Likewise, special attention should be paid to existing stakeholder structure, and potentially existing local ecological knowledge and local institutional arrangements for maintaining wetlands. Such institutions may constitute a basis for building wetland management institutions that have already gained socially acceptable at the local level, in contrast to governmental regulations imposed in top-down fashion.

In summary, this project has shown that both the DP-S-I-R auditing framework and an analytical approach focused around the concept of functional diversity and the related policy objective of maximum functional value diversity are technically feasible and practically relevant for assessing the value of nature conservation. The assignment of economic and social values to a range of different wetland functions has also proved to be a tractable problem. Differences in the economic and cultural geography approaches to valuation highlight the need for a mixed methodology which can attempt to determine the appropriate valuation 'contexts' in which the various methods can be deployed and subjected to an acceptable testing protocol.

6. Conclusion and recommendation for practical implementation of a Decision-Support System for Biodiversity Action Plans (BAPs)

The steps presented in this report towards the development of a holistic integrated framework for environmental indicators are part of an integrated system aiming at the provision of transparent, meaningful and useful information. This system can support and link decision-making at different spatial and time scales with the objective of fostering the protection and sustainable management of natural resources.

Focusing on environmental and social systems and their interactions simultaneously means that the corresponding indicator sets essentially provide the basis for a multi-criteria decision-support framework. Depending on the monitoring scale, in principle the relevant social and environmental effects of decisions can be analysed and evaluated simultaneously. Obviously, this has important consequences for the evaluation of BAPs in terms of single (function) sites and their role at the landscape ecology level. As discussed earlier, the 'value' of such sites can only by properly addressed when the site is viewed within the larger landscape ecosystem and its contributory value is recognised. The 'natural areas' approach and the 'regional profiles' within the Biodiversity Action Plan championed by English Nature fit the overall systems perspective which is recommended in this report.

Comprehensive assessment of ecosystems requires the analyst to undertake the following, not necessarily sequential, steps:

1. to determine the causes of ecosystem degradation/loss, in order to improve understanding of socio-economic impacts on ecosystem processes and attributes
2. assess the full ecological damage caused by ecosystem quality decline and/or loss
3. assess the human welfare significance of such changes, via determination of the changes in the composition of the ecosystem, evaluation of ecosystem functions, provision of potential benefits of these functions in terms of goods and services, and consequent impacts on the well-being of humans who derive use or non-use benefits from such a provision
4. formulate practicable indicators of environmental change and sustainable utilisation of ecosystems (within the DP-S-I-R framework)
5. carry out evaluation analysis using monetary and non-monetary indicators (via a range of methods and techniques, including systems analysis) of alternative ecosystem change scenarios
6. assess alternative ecosystem conversion/development and conservation management policies
7. present resource managers and policy makers with the relevant policy response options.

A number of important aspects of any economic assessment of an ecosystem are presented in Box 6.1.

Box 6.1 Economic assessment of ecosystems

- **Problem orientation**

Any analysis should take account of the prevailingpolitical economy context, equity issues and possible 'stakeholders'. Data inadequacies must be acknowledged and recommendations made conditional upon these limitations.

- **Typology**

A useful common terminology regards **functions** as relationships within and between natural systems; **uses** refer to use, potential use. And non-use interactions between human and natural systems; and **values** refer to assessment of human preferences for a range of natural or non-natural 'objects' and attributes.

- **Thresholds and ecosystem change scenarios**

Thresholds relate to the scale and frequency of impacts. Their occurrence can be presented in a simple three part classification: no discernible effects; discernible effects; discernible effects that influence economic welfare.

- **Ecomomic valuation**

Three broad settings for understanding the ecosystem approach are: impact assessment; partial analysis; and total valuation. For each function or impact, a number of techniques exist for attributing economic value to environmental benefits. Systems analysis and multi-criteria evaluation methods can complement economic cost-benefit analysis.

- **Scale**

The catchment should be the minimum spatial unit for assessing ecological variables, with possible zoning within this. In terms of benefit estimation, the minimum scale is determined by the relevant population affected by any impacts. Temporal scale of analysis is also fundamentally important.

- **Transferability (spatial and temporal)**
- **Transferring scientific results and of economic benefits is problematic. Accuracy of benefits transfer may be improved if based on scientific variables divided into separate components depending on processes, functions and 'state variables'.**

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Annex A. Developing indicators of ecosystem integrity

Landscape Indicators

Introduction and definition

Landscape refers to a mosaic of heterogeneous land forms, vegetation types and land uses (Urban *et al.*, 1987). From this the concept of 'landscape ecology' arose as an approach to introduce ecological principles within land management (Schreiber, 1990). The central focus of this new approach to ecology is described as the interrelationships between landscape structure (including composition), i.e. the patterns of ecosystems across space, and landscape functioning, i.e. the interactions of flows of energy, matter and species within and among the ecosystem structure (Kupfer, 1995). As such, landscapes can be seen as open systems influenced by geomorphology, i.e. the arrangement and differentiation of landforms and the processes that have been or are shaping them, and pedology, i.e. the processes involved in soil formation (e.g. Gerrard, 1992).

Levels of analysis

Mitsch (1992) demonstrated that controls on wetland ecosystem form and function operate at widely different spatial scales and that wetlands have different functions depending on their position in the landscape (e.g. geomorphic position, upstream-downstream or steep versus flat terrain).[1] The spatial scale of a regional landscape may vary from the size of a national forest or park to the size of a physiographic region or biogeographic province. Sizeable geographic regions resulting from geomorphology and climatic regimes have characteristic wetland forms and vegetative types. Some regions contain isolated basins that are reached by vertebrates through their own efforts and are potentially affected by loss or degradation of individual basins, while others form a contiguous habitat and may be affected by reduction or fragmentation (Weller, 1988).

Noss (1983) introduced the term 'regional landscape' to emphasise the spatial complexity of regions. Regionalisation, i.e. the process of defining regions in which biophysical processes are expected to be similar, is a concept developed in the 1970s to understand and classify landscapes using spatial patterns of relevant environmental variables.[2] It is now considered to provide an effective biophysical framework facilitating the assessment of natural biophysical conditions to manage natural resources (Perry and Vanderklein, 1996).[3] For example, water conditions cannot be separated from controlling influences of the surrounding landscape.

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[1] Mitsch *et al.* (1994) distinguish between two hierarchical levels of analysis: the ecosystem level which covers several km^2 spatially and the landscape level which covers hundreds of km^2.

[2] Austin (1972) developed one of the earliest so-called 'ecoregional' maps to provide a geographic basis for making national and regional level management decisions about agricultural concerns.

[3] River basin management is a more familiar term in water resources management than ecoregional management. River basin management has become synonymous with water resources management. It is accepted that basins share similar properties and concerns and represent a logical framework from which to manage. The dominance of river basin management reflects the historical dominance of the linear, upstream-downstream orientation of hydrologists and managers. It is less suited to management of nonlinear lakes, groundwaters or wetlands (Perry and Vanderklein, 1996).

According to Urban *et al.* (1987), landscape assessment, by definition, requires the utilisation of principles that are dictated by the hierarchy of scale and the controlling external environment of the system. Bailey (1987) calls this controlling external environment 'forcing conditions' and uses the concept of forcing conditions to establish 3 major levels of scale to map ecosystems: the macroscale (regional) dictated by climate, the mesoscale (landscape mosaic) dictated by land surface form and the microscale (landscape elements) delineated by vegetation characteristics.

Irrespective of which ecoregional classification is chosen, the classification must include landform type (Klopatek, 1988). Landforms such as floodplains or river channels determine the boundary conditions controlling the spatial location and rate of geomorphic processes which have direct and often predictable effects on ecosystem processes (Swanson *et al.*, 1988). The ecoregional boundaries scale provides the transitional medium to the next level, i.e. wetland type for example, as defined, for example, by Cowardin *et al.* (1979). In this way, the wetland is looked upon as a landscape element.

Landscape indicators

Many variables combine to give a region its peculiar character. Different processes matter in different regions. Consequently, defining a region can be considered an integrative process using many different physiographic and biological variables (Perry and Vanderklein, 1996). Gallant *et al.* (1989) explain that the selected variables may either cause regional variations (for example, climate, mineral availability, physiography) or integrate causal factors (for example, soils, vegetation, land use). In the former case, climate determines for example the relative abundance and seasonality of precipitation and mineral availability the characteristics of water chemistry including salinity and phosphorous levels. However, maps of causal factors alone often do not adequately indicate how these factors interact to determine, for example, water quality parameters. In the latter case, land use maps serve for example as indicators of spatial changes in natural environmental characteristics and resource quality. Land use provides key subtleties (implying important consequences for water resource character and sensitivity to degradation) missed when measuring only climate, topography and physiography. Spatial patterns of past and current land use have for example a major influence on water quality sensitivity and land use maps provide important indicators of those spatial patterns (Gallant *et al.*, 1989).

Topological aspects such as ecosystem size, shape, connectivity and distance between ecosystems may have significant impacts on overall ecosystem functioning (e.g. Swanson *et al.*, 1988; Klopatek, 1988; Johnston, 1994). For example, loss of connectivity within and between wetlands through landscape fragmentation can significantly affect nutrient processing (Johnston, 1994) or the integrity of species communities (Noss, 1983 and 1987; Merriam, 1991). Furthermore, according to Johnston (1994) the cumulative loss of wetland function may or may not be linearly proportional to the area lost. Initial losses of wetland areas may have smaller effects on wetland function than later losses. For example, losses from watersheds containing 10 to 50 percent wetlands have little effect on flood flow, but losses from watersheds containing less than 10 percent have a large effect on flood flow. A similar 10 percent threshold was found in the susceptibility of wetlands to increased loadings of suspended solids (Johnston, 1994). Hence, the impact of habitat fragmentation within the landscape is not only an alteration of size and isolation of a given ecosystem patch, but the restriction of movement of energy, matter and species in and across ecosystems. It is therefore important to be aware of the potential importance of landscape factors which might influence ecosystem functioning, including edge effects, corridor functioning and boundary dynamics (Kupler, 1995).

Water Regime Indicators

Introduction and definition

Water regime is central to the definition of many ecosystems, especially wetlands. Water enters wetland systems by a number of mechanism ranging from river influx, tidal inundation, land surface run-off, groundwater discharge and precipitation. Similarly, water may leave a wetland through a river flow, groundwater recharge, tidal ebbing and evapotranspiration. These flows are, by and large, extremely variable and are stochastic in nature. The storage of water within a wetland system is determined by the balance of inflow and outflow relationships and the topography of the underlying basin. Storage within the main body of the wetland is very much dependant on landscape features such as the volume of accumulated sediment, transfer rate of water through the system and depth of adjacent water bodies. In fact, some wetlands may be flooded for extended periods of time whilst others may only be flooded for short and infrequent intervals.

Water acts both as a stimulus and a limit to species composition and richness in ecosystems, depending on water storage and physical hydrodynamics. Alterations to hydrological conditions can have a significant non-linear impact on species composition and diversity, productivity, the exchange of organic material (import-export) and nutrient cycling within the wetland (Mitsch and Gosselink, 1993). Surprisingly, of the many thousands of vascular plant species in existence relatively few have adapted to water logged soils and even fewer to water logged saline soils. A high water table or significant surface floodings act as a selective pressure to support vegetative communities often tolerant of anoxic soil conditions. Because of this, water logged soils generally support a lower species richness than less frequently flooded systems, but also an abundance of rare niche specialists.

Levels of analysis

Taking wetlands as an example ecosystem, their classifications are generally based on ecology, physical and geomorphological characteristics. One of the first widely used classification systems, devised by Cowardin *et al.*, (1979), divides wetlands into marine, estuarine, riverine, lacustrine and palustine, with associated sub-classes, based on simple hydrological and geomorphological criteria. Wetland systems portray heterogeneity across a range of spatial scales from extensive landscape features down to local water table patterns and levels of water quality.

Measurement of material (nutrient, sediment and biotic) budgets or mass balances between input and outputs, which take place on a range of spatial and temporal scales, are important in characterising water regime. Quantifying, through mass balance calculation, the flow of nutrients through an open system such as a wetland is no simple task. Kadlec and Knight (1995) point out that a proper mass balance must satisfy the following conditions:

1. The system for the mass balance must be defined carefully. The actual definition of the system is dependant on the context of the study, though a global term might be applied when the entire water body of a wetland is investigated. Alternatively, if only

a section or particular element of the system is under investigation an internal mass balance might be computed.

2. The time period for the inputs and outputs must be specified. Fluxes vary across a range of temporal scales from short term tidal, to seasonal or even longer climatic event cycles. The finite period of evaluation must be put into context with these longer term cycles if relatively accurate flux calculations are to be determined.
3. All the inputs and outputs to the chosen system must be included. The concept of mass balance should be invoked to calculate one or a group of material fluxes.
4. Any production and destruction reactions taking place within the wetland must be identified to take into account material changes.
5. Waterborne constituent flows are determined by separate measurements of water flow and concentrations within the flow. As such accurate water mass balance is a prerequisite to an accurate material flux mass balance.
6. Where possible it is desirable to demonstrate closure of the mass balance.

A number of studies have been undertaken to calculate water and constituent material fluxes on a range of scales from a range of environments (Mitsch (1994) includes a number of examples), but the controversy over whether certain wetlands actually are sinks or sources of various materials reflects the uncertainty in these studies.

Hydrological indicators

Utilisation of a strictly hydrological approach to evaluating wetland ecosystems has its limitations as direct evidence of hydrology is often difficult to obtain and may require monitoring over several months to identify seasonal variations. In many countries monitoring of baseline data such as regional rain fall or tidal level exists, but care must be taken when extrapolating general regional data to a local site specific situation. For instance, considering coastal wetlands, tidal elevation data is available at many sites around the UK coastline. However, because of the effects of estuarine morphology and, locally, saltmarsh morphology on the progression of a tidal wave, extrapolation of a tidal elevation from a tidal station to a marsh elsewhere in the estuary is prone to considerable error. Likewise in fluvial systems, a complex network of wetlands act to store water and defuse flood events. Thus, water elevations on one flood plain need not equate to water levels elsewhere in the catchment. For extrapolation of information to be meaningful, a programme of detailed monitoring is required to provide inter-site correlation.

With these complexities in mind, it is rarely possible to find hydrological data specific to wetland systems, unless it has been subject to a specific site investigation.

Fortunately, hydric soils and hydrophylic vegetation are reliable indirect indicators of wetland hydrology and can be used to infer its presence when the hydrology has not been altered. When hydrology has been altered, soil and vegetation might not be reliable indicators, and the hydrological status must be evaluated independently (National Research Council, 1995). Hollis and Thompson (1998) point out that in hydrological studies of wetlands it is not unusual to gather data from non-standard sources such as a farmer's records, hydrological data from non-hydrological agencies, oral information from local people or photographic evidence. A major technical achievement will be to determine an average or characteristic water regime for sites on which there is no hydrological data, or for which hydrological data only covers a short time interval (National Research Council, 1995).

Many reviews acknowledge that the key hydrological factors underpinning the existence of a wetland are the rate and balance of water supply and loss and water storage (Hughes and Heathwaite, 1995). These factors control the variations in wetland character in terms of position of the water table and frequency of flooding events (Heathwaite *et al.* 1993). In a discussion of hydrological analysis requirements, Hollis and Thompson (1998) define six key hydrological variables which require monitoring for effective wetland management: water level regime, land-area-volume relationships, water balance, turnover rate, extremes and water quality.[4]

Water quality represents the compositional aspect of the water regime. Water richness in nutrients, oxygen, toxicants and pH levels and clarity are important in influencing wetland ecology and functioning. Many natural external and internal factors influence catchment water quality, including self-regulatory systems within wetlands, which act to recycle nutrients and contaminants (Mitsch and Gosselink, 1993). Increasingly though, anthropogenic factors are determining the quality of water in wetland systems. These forms of pollution may come from upstream discharges of chemicals or nutrients from outfalls and from non-point pollution sources such as run-offs from agricultural land or otherwise developed areas.

A number of critical water quality impacts result from different land uses: change in suspended sediment load, organic matter and biochemical oxygen demand, bacteria, parasites and viruses, nutrient loads, heavy metals, organic toxins such as pesticides and hydrocarbons, acidification, salinisation and temperature (Perry and Vanderklein, 1996). The effects of pollution on the long term sustainability of biota is not so well understood and so creation of threshold toxicant limits is difficult to ascertain, particularly if multiple pollutants are involved. The reason for this is that pollutants may act to increase the death rate or to reduce the birth rate, and up to a certain point these effects need not lead to continued population decline. Species populations may therefore withstand a certain amount of extra mortality, reduced reproduction, without declining in the long term (Newton, 1988).

Biodiversity Indicators

Introduction and definition

Biological diversity was abbreviated into biodiversity in the mid 1980s to capture the essence of research into the variety and richness of life on Earth (Jeffries, 1997). This variety of life can be studied at different levels: genetic variation, number of species or ecosystems. Nowadays, biodiversity is understood as consisting of more than just species diversity, although in practice it is still commonly measured by counting the number of species in an area and the turnover of species among areas (Williams and Gaston, 1994).

Biodiversity can be defined and measured from the perspective of different scientific disciplines such as systems ecology or biology at different hierarchical, spatial and temporal scales. In the two most widely cited definitions, by the American Office of Technology

[4] In this study we have separated the compositional properties of water quality from the structural hydrological characteristics.

Assessment and the Biodiversity Convention[5], biodiversity comprises both variety and variability and covers both biotic and abiotic complexes. The inclusion of the word variability besides variety or diversity among living organisms and the emphasis on both biotic and abiotic components in both definitions explicits the difficulty to pinpoint the concept of biodiversity to a single scale of measurement and introduces at the same time a strong analytical aspect to its measurement.

Levels of analysis

Three main biodiversity monitoring categories or levels can be distinguished in literature (e.g. Nos, 1990): genetic diversity, taxonomic diversity and ecological diversity. These categories originate from distinct 'schools' or disciplines within biology and systems ecology with different emphasis or focus on biodiversity as the outcome of evolutionary and ecological processes (Jeffries, 1997). The ecological domain of biodiversity focuses mainly on populations, communities or ecosystems and their interactions with the physical environment, whereas the evolutionary domain can be defined by a focus on genetic processes and patterns and the variation they create.

Genetic and subcellular diversity includes all biodiversity expressed within individual cells plus non-cellular organisms such as viruses. Genetic variation may be driven by the physical environment, but genetic diversity is the main focus point linked to species since genetic processes could function without any environmental influence. Without genetic variation, evolution and adaptation cannot occur (Jermy *et al.*, 1995). Even when a habitat has been preserved, evolution is still going on, in the form of longer term adaptation to climatic or other environmental change or on a shorter time scale. Even in the best preserved habitats ecological communities are dynamic entities with species interacting with each other. The measurement of genetic diversity is a rapidly evolving area (Jeffries, 1997).

Taxonomic diversity refers to species diversity and is the most popular idea behind biodiversity. Taxonomy is the theory and practice of describing the diversity of organisms and the arrangement of these organisms into classifications. The most fundamental unit for taxonomic research is the individual organism, while the species is widely accepted as the most basic of natural units. However, there exists a lot of disagreement regarding what a species exactly is. Two main concepts exist, which differ in emphasis on the reproductive process and genetic correlation (Jermy *et al.*, 1995). The first one defines species as '*groups of interbreeding natural populations isolated from other such groups*' (Mayr, 1969), while the second one defines species as *'the smallest aggregation of populations diagnosable by a unique combination of character states in comparable individuals*' (Nixon and Wheeler, 1990). Given the definition of biodiversity in the Biodiversity Convention (see footnote), the latter seems to be a more appropriate definition of species, since the former excludes a large number of organisms, particularly plants, which reproduce asexually.

After species have been defined, their classification based on a hierarchical pattern of genetic relationships or descendance is an important next step for the purpose of extrapolation and

[5] The Office of Technology Assessment (OTA) defines biodiversity as '*the variety and variability among living organisms and the ecological complexes in which they occur*' (OTA, 1987). The Biodiversity Convention at the UN Conference on Environment and Development in Rio de Janeiro in 1992 defines biodiversity as '*the variability among living organisms from all sources including terrestrial, marine and other aquatic ecosystems and ecological complexes of which they are a part: this includes diversity within species, between species and of ecosystems*'.

estimation of the distribution of species richness more widely. Cladistic analysis or cladograms are widely accepted as providing an efficient method for representing information about organisms and seem to be regarded as having a firm foundation for establishing relationships between different organisms. A cladogram is based on 3 main assumptions (Farris *et al.*, 1970; WCMC, 1992): (1) features shared by organisms form a hierarchic pattern; (2) this pattern can be expressed as a branching diagram; (3) each branching point symbolises the features held in common by all the species arising from that node.

Besides the existence of a hierarchy in taxa based on genetic relationships or descendence, another hierarchy can be found in the feeding interactions between the biotic components of an ecosystem (also called ecological community) as part of the food web (e.g. Putman, 1994). A food chain is an energy path or linear sequence of links that depicts who eats whom or what. Primary producers (or *autotrophs*) comprise the first trophic level. These species (mainly green plants but also algae and cyanobacteria in some ecosystems) are responsible for trapping solar energy and converting it into chemical energy and tissue biomass which may then be utilised by the rest of the ecosystem. All other members of the community are dependent, either directly or indirectly, on the primary producers for energy. Because some energy is used at each trophic level and the transfer of energy between trophic levels is never completely efficient, less energy is available at higher trophic levels.

Consumers (or *heterotrophs*) are animals and micro-organisms (and occasionally plants) that feed on primary producers and each other. The second trophic level is made up of primary consumers (herbivores which feed directly on primary producers), while secondary consumers (carnivores which feed on herbivores) comprise the third trophic level and tertiary consumers (carnivores which feed on other carnivores) comprise the fourth. Omnivorous animals may be given partial representation in several trophic levels in proportion to the composition of their diet. Decomposers (also classified as *heterotrophs*) are the species which feed on and break down dead plant and animal material making the component nutrients available to the system again. Understanding the flow of energy through an ecosystem is important for conservation and any sustainable exploitation of that system (Jermy *et al.*, 1995).

Biodiversity indicators

Biodiversity indicators are being compiled in massive numbers despite arguments that they are answers to questions which have not yet been articulated (e.g. Norris and Norris, 1995). Monitoring of biodiversity usually occurs at species level. Higher taxa can for example be used as an indicator to predict overall biodiversity. The use of selected species groups as indicators of overall biodiversity is attractive because if suitable indicator relationships can be shown to exist sampling for just the selected species will greatly reduce survey costs. However, too often evidence is lacking that there exists a relationship between the indicator group and overall biodiversity (Williams and Gaston, 1994).

Adequate definition of the spatial and temporal scales on which diversity is measured is of paramount importance (Rosenzweig, 1995). For example, species-area curves support the rule that more species will be found if a larger area is sampled. Seasonal variation and corresponding migratory patterns of for example birds, on the other hand, cause diversity to oscillate through the year. The relationship between succession after natural or human induced disturbances and diversity is rather unclear. Some empirical research exists (see

Rosenzweig (1995) for an overview), but often not more than a few years were spent on monitoring communities.

Plant and animal species have also been used as indicators of environmental conditions such as water quality (Norris and Norris, 1995) or habitat quality (e.g. Weller, 1988). Most uncertainty seems to exist in establishing their validity, that is, that they actually measure those features of the environment one is interested in and that they change in a meaningful way with respect to environmental change. However, given the complexity of natural systems, the probability is small that a single species can serve as an index of the structure and functioning of an ecological community or even an entire ecosystem (Ward, 1978; Cairns, 1986). Assuming that the relevant species-habitat relationships are adequately modelled, population density has been used as an indicator of habitat quality for that species or one or more species have been used to indicate habitat suitability for other species. Landres *et al.* (1988) identified several problems when using indicators to assess habitat quality. Weller (1988) argues that in order to derive meaningful habitat patterns and impact assessments it is essential to understand water regimes, vegetation patterns and vertebrate habitat strategies. A single measure may be highly misleading given that the structure of vegetation and communities is dynamic.

Noss (1990) lists 5 categories of species that may warrant special monitoring or protection:

1. Ecological indicators: species that signal the effects of perturbations on a number of other species with similar habitat requirements.
2. Keystones: pivotal species upon which the diversity of a large part of the community depends.
3. Umbrellas: species with large area requirements which if given sufficient protected habitat area will bring many other species under protection.
4. Flagships: popular, charismatic species that serve as symbols and rallying points for major conservation initiatives.
5. Vulnerables: species that are rare, genetically impoverished, dependent on patchy or unpredictable resources, extremely variable in population density persecuted or otherwise prone to extinction in human-dominated landscapes.

Keystone species are identified by Jermy *et al.* (1995) as playing a crucial role in an ecosystem. Examples of groups which include keystone species are predators, parasites and pathogens which help to maintain population levels of prey and host species, large herbivores and termites which control ecological succession, species that create and maintain landscape features such as waterholes and wallows in arid areas, pollinators, seed dispensers and other obligate mutualists, and plants that provide a resource in time of scarcity.

Some keystone species are essential for the formation of the biotope in which they live or ecosystem functioning. Closely related to this is the idea of guilds, i.e. functional groups or clusters of species interacting among themselves more strongly than with other elements of the community (e.g. Keddy, 1990; Putman, 1994). These species perform a similar function in a given ecosystem.

Regarding this latter point, three competing theories as to how an ecosystem might respond to loss of species diversity exist (Jermy *et al.*, 1995). The 'redundant species' hypothesis suggests that there is a minimum set of species required for an ecosystem to function and that adding or losing others does not affect processes (Walker, 1992; Lawton and Brown, 1993).

The 'rivet hypothesis' (Ehrlich and Ehrlich, 1981) suggests that all species are essential and that as species are lost, functioning is impaired. The 'idiosyncratic response' hypothesis, finally, suggests that functions change when diversity changes, but in an unpredictable way (Vitousek and Hooper, 1993; Lawton, 1994).

Experimental tests of these hypotheses in controlled laboratory conditions using a controlled environmental chamber (ecotron), suggests that most processes (decomposition rates, nutrient uptake etc.) vary idiosyncratically with species richness, but that both uptake of carbon dioxide and plant productivity declined as species richness declined as predicted by the rivet hypothesis (Naeem *et al.*, 1994 and 1995). As hypothesised in systems theory, it seems that a greater number of links in a food web will provide more opportunities for checks and balances should any environmental change occur.

Annex B. Indicators of ecosystem integrity – Norfolk Broads wetlands case study

Table B1: Landscape indicators

Ecosystem Attribute	Indicator	Purpose/objective	Relevance for sustainable ecosystem management
Structure	1. Area size	• indicate the administrative-political and natural boundaries of the ecosystem management area • assess the overall trend in area loss over time of specific ecosystem forms (in combination with composition indicators) • assess viability of area size dependent species communities	• effect of cumulative ecosystem area loss on different ecosystem functions (e.g. Johnston, 1994) • minimum area size required for species richness (island biogeography) (e.g. Brown and Dinsmore, 1986)
	2. Connectivity	• indicate the habitat structure of the ecosystem • assess the relative ease of species movement • assess the overall integrity and resilience of the ecosystem • assess impairment of overall ecosystem functioning	• all biological and ecological systems have a degree of resilience; they will tolerate a certain level of stress or depredation, while maintaining the capacity to recover; even if individual elements of a system are destroyed, these elements can often be restored provided that the essential network of relationships that constitutes the system remains (e.g. Clayton and Radcliffe, 1996) • loss of connectivity between ecosystems through landscape fragmentation can significantly affect the cumulative function in nutrient processing (Johnston, 1994) • effect of fragmentation on the integrity of the species community (Noss, 1983, 1987; Merriam, 1991)
	3. Proximity	• indicate the relative distribution of ecosystem and landscape types • assess the opportunity for species migration • assess the possible influence of ecosystem sites on each other within and between ecosystems through consideration of the distance between sites	• reserves which are more isolated may experience lower rates of species immigration (Diamond, 1975); over time this may lead to a decline in species pool within an isolated system as extinctions occur • marginal areas between ecosystems are often characterised by unique transitional species communities
	4. Shape	• indicate the relationship between edge length and area of an ecosystem • assess possible implications of edge effects on ecosystem integrity • assess implications of boundary dynamics on species composition	• highly convoluted ecosystems may be more susceptible to disturbance or stress because of proportionally larger edge region and exposure to external pressures (Schonewald-Cox and Bayliss, 1986; Schonewald-Cox, 1988) • increased soil surface area in proportion to volume may increase rate of nutrient recycling and attenuation of hydrodynamic energies
	5. Flood plain location	• assess extent of transition zone between uplands and aquatic system • assess wetland functioning as flood control buffer	• effectiveness of wetland to flood control increases when ecosystem is located further downstream (Ogawa and Male, 1983)

Table B1: Landscape indicators (continued)

Ecosystem Attribute	Indicator	Purpose/objective	Relevance for sustainable ecosystem management
Composition	1. Landform	• indicate the natural boundaries of ecosystem management sites • determine ecosystem type or class • indicate the spatial location and distribution of impact assessment and the relevant biogeochemical processes involved • assess the heterogeneity of the ecosystem and hence the system's suitability as habitat for ecologically sensitive species or species that utilise multiple habitat types	• landforms determine the boundary conditions controlling the spatial location and rate of geomorphic processes which have direct and often predictable effects on ecosystem processes (e.g. Swanson *et al.*, 1988) • landscape features such as heterogeneity and related features of landscape composition (proportions of particular habitats) can be major controllers of species composition and abundance and of population viability for sensitive species (e.g. Noss and Harris, 1986)
	2. Vegetation cover	• indicate vegetation composition (at community level, not individual species level) • identify and describe individual communities and the relationship among communities • describe ecosystem cover type (standardisation enables comparison among disparate monitoring efforts) • global indicator of hydrology, water availability and water quality • assess the harvest potential of wetland sites • indicator of biomass and resource productivity • global indicator of species communities	• monitoring the positions of ecotones at various spatial scales may be useful to track vegetation response to climate change and disruptions or disturbance regimes (Noss, 1990) • strong relationship between frequently recurrent or sustained soil saturation (with water) and the development of communities dominated by plants specifically adapted for or requiring such conditions (hydro-phytic vegetation); communities composed of these plant species have been used for decades to identify wetlands for example (CCW, 1995) • repeating expensive inventories of species distributions can be impractical for monitoring purposes; periodic inventories of vegetation can effectively monitor the availability of habitats over broad geographic areas; inferences about species distributions can be drawn from such inventories (Noss, 1990)
	3. Soil characteristics	• assess effect of soil texture on ecosystem functioning such as nutrient recycling or sediment retention • assess effect of chemical soil composition on ecosystem functioning	• floodplain soils exhibit characteristics of both sediment transport and deposition and soil formation (Gerrard, 1992) • impact of nitrogen and phosphorous on eutrophication
Functioning	1. Land use changes	• assess land use changes over time and hence provide insight in possible stress and disturbances • global indicator of landscape processes of change	• although there is no conceptual basis for fine-scale predictions of responses that can be generalised, there are numerous general principles that predict the direction and to some degree the magnitude of ecosystem responses to a wide variety of disturbances such as road construction, agricultural practices etc. (Risser, 1988)

Water Regime Indicators

Selection of water regime indicators

In table B2three of the seven hydrological indicators proposed by Hollis and Thompson (1998) have been adopted, i.e., water level regime, level-area-volume relationships and water balance, to reflect the structural component of the wetland system. Composition basically reflects water quality and has been broken down into load components (sediment, organics, nutrients, toxicants, oxygen and organisms). Finally, turnover is suggested as an indicator of wetland hydrology processes.

Biodiversity Indicators

Selection of biodiversity indicators

In this paper, we will continue the traditional focus on species level in the compilation of biodiversity indicators, amongst others because of the time and costs involved in monitoring genetic variety, but complement these indicators at the same time with relevant environmental and ecological habitat variables. This corresponds to the approach outlined by Noss (1990), whose break-down of biodiversity in structural, compositional and functional biodiversity has been used as the overarching framework for ecosystem integrity assessment in this paper.

In view of the importance of food chains in ecosystems to their structure, composition and functioning, the food web will be the central concept in the compilation of biodiversity indicators. This means that trophic levels will provide the hierarchy within this indicator sphere, while the physical environment establishes the limits to food chains and food webs as measured in the previous sections in terms of landscape and water regime indicators.

In order to determine the consequences of environmental impacts on food webs, the structure of ecosystem food chains has to be determined first (the structural attribute of biodiversity in terms of trophic levels) and secondly the impact of environmental variability (for example through human disturbance) on the food web. In terms of habitat and food support, ecosystem size, shape, connectivity and distances between ecosystems need to be measured to evaluate potential cumulative impacts (Klopatek, 1988), issues which are addressed in the landscape section. Briand (1983) found that for a given number of species, the trophic linkage (connectance) in the food web is significantly lower in variable than in constant environments, i.e. habitats in constant environments possess longer food chains whereas those with variable environments have greater widths.

Using one or more vertebrate to indicate environmental conditions or habitat quality (for that particular and other species) has to be based on an established valid relationships between species and the environmental conditions or habitat features of interest. Indicator species often have told us little about overall environmental trends and may have deluded us into thinking that all is well with an environment simply because an indicator is thriving (Noss, 1990).

Landress *et al.* (1988) address the criteria used in the past to select species indicators for ecological or habitat assessment. From these, we will adopt the following three for the purpose of assessing overall ecosystem integrity:

- Area requirement: if a single species is used as an indicator of habitat quality or of a community, it is commonly assumed that the species should require a large area for its territory. Ideally, the indicator would have greater area requirements than any other species in the community, because the larger area required, the more likely it is to include the spectrum of resources needed by other organisms dependent on that particular habitat. These species are usually called umbrella species (see above). Brown and Dinsmore (1986) found empirical evidence for the existence of minimum wetland size for particular bird species (5 ha). As wetland size increases, the number of individuals and species increase in some pattern, often a sigmoid curve, as well. However, Martin and Karr (1986) showed that diversity has a closer relationship to the amount of food resource present than to area alone, another reason for making the food web a central concept in the biodiversity indicators sphere.

- Specialists: species vary in the range of resources or habitats used, each species falling somewhere along a specialist-generalist continuum. Odum (1971) suggested that specialists are better indicators because they are more sensitive to habitat changes. Sensitivity to environmental conditions obviously is a necessary prerequisite to the use of species as indicators of environmental change. In wetlands, amphibians have been suggested by Keddy *et al.* (1993) as sensitive to a wide range of contaminants given their semipermeable skin and the fact that they inhabit both aquatic and terrestrial habitats. However, their reliance on more than one habitat and hence exposure to a variety of possible disturbances or contaminants makes it at the same time increasingly difficult to associate their decline with one or more specific environmental condition or feature.

- Residency status: since non-permanent or migratory species are subject to a variety of sources of mortality, it may be tricky to use them to indicate environmental conditions on-site. A decline in their abundance may be unrelated to habitat conditions on the breeding or foraging grounds. Permanent (not necessarily endemic) residents will hence usually be more reliable indicators. However, also in the case of using permanent residents, it is important to take into account the spatial and temporal scale on which species density is measured such as the openness of the ecosystem and succession stages (see for example Rosenzweig, 1995) and population dynamics such as generation times or life cycles and the impact of changing environmental conditions on evolutionary change between generations (see for example Begon and Mortimer, 1992).

Summarising, umbrella and keystone species will be identified to indicate (1) overall ecosystem biodiversity, (2) ecosystem habitat quantity and quality and (3) environmental conditions.

Finally, the functional biodiversity component will be indicated by biomass. In wetlands, biomass appears to be an excellent indicator of trophic status (Keddy *et al.*, 1993), although Onuf and Quammen (1986) and Klopatek (1988) conclude that the primary productivity of a wetland (i.e. the total amount of energy captured within a community by the producer level) is not a critical variable in determining wetland food chain support. On the other hand, Odum (1975) suggested that variation in type, quantity and quality of energy flows through communities may have profound implications for its whole structure and functioning given possible links between energy flows and diversity. DeAngelis (1980) furthermore observes that inherently unstable structures may in practice prove stable if energy flows through the system are high enough. Putman (1994) considers 4 aspects of resource flows through communities: the amount of energy handled by the system, the efficiency of energy transfer between trophic levels , the rate or speed of energy flow and the nature of associated nutrient cycles (open or closed, sedimentary or non-sedimentary). Differences in the efficiency of energy transfer at different trophic levels offer insight into the different mechanics and efficiency of operation at these levels. Keddy *et al.* (1993) propose the use of plant biomass as a performance indicator of primary producers since it is not only relatively easy to monitor, but may also provide more information than the biomass of a few selected species in view of the response time to the cumulative effects of nutrients available over a long period of time.

Table B2: Water regime indicators

Ecosystem Attribute	Indicator	Purpose/objective	Relevance for sustainable ecosystem management
Structure	1. Water level regime	• describe the temporal variation in surface and sub-surface water level and reflect the relationship between water flux and wetland topography	• as diffusion of oxygen through aqueous solution is of the order of 10,000 times slower than through a porous medium, such as a drained soil, the position of the water table within a wetland is important in determining the microbial respiration and porewater geochemistry (Gambrell and Patrick, 1978), and hence macrophyte production • integral component of material flux calculations • depth of surface water influences availability of light to primary producers
	2. Level-area-volume relationships	• describe the area flooded by waters and determine the geographical coverage of the wetland	• changes in LAVR impacts geographically on species composition and richness (Hughes and Heathwaite, 1995) • essential to understand how hydrological input and outputs impact on ecological elements such as species distribution
	3. Water balance	• indicate the interactions of precipitation, river flooding, groundwater discharge, tidal exchange, evapotranspiration, water abstraction, catchment management and topography on other hydrological characteristics	• frequency of inundation is a determinant of species composition and richness (Wheeler and Shaw, 1995; Hughes and Heathwaite, 1995) • influences the budget of dissolved and particulate material across the wetland/catchment interface (Murray and Spencer, 1997) • calculation of water balance is an important component of material budget calculations • a positive water balance is essential for carbon preservation in peatlands (Mitsch and Gosselink, 1993)

Ecosystem Attribute	Indicator	Purpose/objective	Relevance for sustainable ecosystem management
Composition	1. Sediment load	• indicate amount and nature of sediment supplied or lost from a wetland system • determine rate of sediment supply	• sedimentation rate determines topography-water level interactions in wetland systems; • important in determining long-term resilience of wetland to pressures such as rising sea-level or subsidence • the nature of the sediment may influence wetland resistance to remobilization by hydrological events and influence nature of biogeochemical interactions; • high sediment loads may be responsible for reduction of primary production of aquatic plants, blockage of fish gills, smothering of benthic organisms; • associated mobilisation of agricultural pollutants
	2. Organic load	• indicate nature and supply or release of organic matter	• organic sedimentation rate influences carbon availability and below ground productivity • possibly influences productivity of nearby ecosystems such as estuaries (Gordon and Cranford, 1994) • microbial breakdown of untreated sewage or decaying vegetation can deplete oxygen levels within the water column, stressing oxygen dependant organisms. • a factor in the development of anoxia within waterlogged sediment influencing biogeochemical processes such as nutrient recycling and absorption of metals • oxygen level important in the reduction of bacterial levels derived from sewage • required as part of carbon flux calculations • if know amount of oxygen required to degrade organic matter can estimate effect on BOD
	3. Nutrient load	• indicate nature and supply or release of nutrients • estimate potential for eutrophication	• wetlands require an influx of nutrients for biomass growth • excess loads of nitrogen and phosphorus may stimulate algal blooms with associated impacts on biodiversity and water quality • nutrients may pose a health hazard in high concentrations

Ecosystem Attribute	Indicator	Purpose/objective	Relevance for sustainable ecosystem management
Composition	4. Toxicant load	• indicate potential for health implications for humans and change in organism diversity within ecosystem	• toxicant levels is one of a number of stress factors which may cumulatively lead to a reduction in species diversity (Stansfield *et al.*, 1989) • although the effects of excess toxic loading are often apparent in mass mortality of organisms it is usually difficult to correlate toxicant load at subcritical levels directly with species abundance because of non-linear relationships and additive effects with other factors and toxicants (Newton, 1988)
	5. Oxygen level	• indicate oxygen availability to dependant organisms • indicate nature of bacterial respiration taking place and hence possible biogeochemical processes • indicate the potential for metal assimilation within wetland soils	• depletion of oxygen can lead to stress on oxygen dependant organism and reduction in species diversity • presence of oxic and anoxic zones within sediment influences nutrient nutrient cycling rate (Armstrong and Armstrong, 1988); • organic decomposition occurs most rapidly under oxic conditions • redox determines mobility of inorganic pollutants, such as metals, thus determining the assimilative capacity of the wetland system and bioavailability of pollutants • rate of oxygen consumption can be used to investigate rate of biogeochemical processes • BOD reflects utilisation of oxygen by respiration and availability to oxygen dependant organisms
	6. Bacteria, viruses and water-borne parasites	• indicate potential health hazards	• globally water-borne diseases and parasites pose perhaps the greatest threat to human health, particularly in the developing world (Nash, 1993) • various species of bacteria are important in the regulation of water quality by assimilating DOC (Cole and Pace, 1995), releasing nutrients during the remineralisation of organic matter (Anderson, 1992) or acting as both source and sink for organic nitrogen (Goldman *et al.*, 1987; Sanders and Purdie, 1998) • bacterial breakdown of organic matter may lead to reduced oxygen levels in water

Ecosystem Attribute	Indicator	Purpose/objective	Relevance for sustainable ecosystem management
Composition	7. Temperature	• determine rate at which chemical reactions takes place both inorganically and biologically mediated	• temperature controls rates of biochemical process, determining factors such as porewater redox, primary productivity, organic decomposition and species diversity (e.g. Newman and Clausen, 1997) • nitrogen flux processes such as ammonification, nitrification and denitrification are all temperature dependant (Kadlec and knight, 1996) • phosphorus absorption to substratum is temperature-correlated due to sensitivity to redox (Nichols, 1983) • an important component of biochemical and geochemical mass balance and flux calculations
Functioning	1. Turnover rate	• describe the rate at which water and constituents are transmitted through the wetland system	• turnover rate determines the throughput rate of water constituents through the wetland • influences biochemical wetland properties such as nutrient accumulation and environmental quality by flushing of pollutants and water-borne organisms (Lent *et al.*, 1997) • may influence rate of sediment supply or removal from wetlands

Ecosystem Attribute	Indicator	Purpose/objective	Relevance for sustainable ecosystem management
Structure	1.Trophic levels of food web	• describe food chain in ecosystem • indicate energy transfer levels	• food chain support important ecosystem function • dependency of species on specific trophic structure • important determinant in calculation of primary productivity • biodiversity has a closer relationship to the amount of food resource present than to area alone (Martin and Karr, 1986)
Composition	1. Key species	• indicate species richness • indicate overall wildlife habitat quality • indicate specific environmental conditions	• keystone species are essential for the formation of the biotope in which they live or ecosystem functioning
	2. Umbrella Species	• indicate minimum wildlife habitat requirements • indicate overall species richness	• empirical evidence for the existence of minimum wetland size for particular bird species (e.g. Brown and Dinsmore, 1986) • the larger the area required, the more likely it is to include the spectrum of resources needed by other organisms dependent on that particular habitat
Functioning	1. Primary productivity	• biomass productivity indicator • energy transfer efficiency indicator	• insight into the different mechanics and efficiency of operation at trophic levels • may have profound implications for whole wetland ecosystem structure and functioning given possible links between energy flows and diversity (Odum, 1975) • inherently unstable structures may in practice prove stable if energy flows through the system are high enough (DeAngelis, 1980)

To summarise, a set of 24 quantitative and qualitative environmental indicators has been taken from the literature to assess the in-impact of pressures on wetland ecosystems. The Organisation of the indicators into three distinct fields, organised according to three different ecosystem components may suggest *independency* between indicators or indicator sets. However, in fact it is the *interdependency* and *compatibility within* and *between* the indicators for ecosystem components and wetland domains across different scales and hierarchies that is of primary interest and provides the main challenge to produce a genuinely integrated indicator framework.

An important next step will be to assess the extent to which this Organisation results in useful and meaningful information in terms of the systems approach set out from the beginning in assessing the sustainability of wetland ecosystem use and management. In other words, how compatible and comparable are the various indicators across scales and to what extent do they complement each other to sketch as much as possible a comprehensive and complete picture while respecting at the same time real resource constraints (time and money) in monitoring efforts. The indicators measure different dimensions of wetland ecosystems and the comparability or amount of '*common ground*' of these different indicators has to be assessed in different case studies.

In practice, many of the indicators may consist of several sub-indicators, depending on the specific wetland site or system of interest, possibly resulting in something like an indicator tree or hierarchy. Also, some indicators will be more relevant than others, again depending on the specific conditions or circumstances prevailing at different wetland sites or within specific systems. Hence, it is important to bear in mind that the indicator sets are in this sense not *absolute* to the overall assessment of wetland ecosystems. Their applicability, meaningfulness and usefulness, especially in relation to each other, in wetland management practices have to be considered carefully in each specific situation.

It is also important to point out that the indicators do not indicate whether or not specific management intervention or environmental change maintains ecosystem integrity. In other words, the presented indicators are not *sustainability* indicators. They reflect certain aspects of a system state which are considered relevant, based on scientific understanding of wetland ecosystems, in order to be able to assess the impacts of human activities or evaluate management intervention in a comprehensive and reliable way. In order to assess sustainable use of wetlands, benchmarks or reference points are needed, which can be partly established based on scientific understanding of for example absorption or carrying capacities of an ecosystem but often include normative judgements towards wetland conservation.

Furthermore, the selection. of specific indicators is arbitrary. Besides scientific significance, important other selection criteria for the indicator sets presented were expected ease of observation and computation and cost-effectiveness. For instance, ideally one would perhaps also look at genetic variety present in wetlands in order to get a comprehensive view of possible wetland degradation, but this will be a time consuming and expensive exercise. Hence our reliance, *inter alia*, on species indicators. On the other hand, very little is known about the exact role of specific species in an ecosystem and to what extent they support specific ecosystem functions. It is expected though that a framework as presented here may perhaps provide further insight into the relationship between species and ecosystem functioning by focusing simultaneously on important ecological, biological and hydrological aspects of the system as a whole. In the presentation of the indicator sets we have also been led to some extent by our own limited knowledge of actual wetland management and monitoring practices.

Finally, most of the indicators presented here are *state* variables related to the *effects* of human intervention on wetland ecosystems. Land use change is the only indicator that directly relates to an environmental pressure. When comparing the outcome of the state indicators with some targeted reference state or situation, insight is needed in the often complex relationship between cause and effect in order to be able to change (reduce) existing pressures on ecosystems. The suggested indicators may furthermore monitor environmental change at different points along a continuum from source to effect. Hence, related to their interdependency, is the question to what extent the *efficiency* of monitoring can be enhanced by focusing on a reduced number of selected points along this continuum when knowing more about the dependency relationships in a given system.

Some actual thresholds and keystone species for the Broads

Given the level of research and monitoring undertaken within Broadland over the last twenty years, and in many circumstances to the turn of the last century, there is a wealth of information to create indicators representing the state of health of the broads ecosystems. For example, currently in relation to eutrophication regular monitoring of a number of water quality parameters (total N, total P) coupled with monitoring of macrophyte, and invertebrate and vertebrate diversity is clarifying nutrient impact on biodiversity.

Below we look at some example indicators in terms of thresholds and keystone species

Critical thresholds

Critical thresholds can be identified for the hydrological structure, biogeochemistry and ecology of the Broads. The area's hydrology, biogeochemistry and ecology are highly interrelated. A change in one can have significant impacts on the other and again result in feedback or knock-on effects. For the purpose of clarity, some examples of hydrological, biogeochemical and ecological thresholds for the broads will be briefly reviewed in separate subsections.

Critical hydrological processes

1) Water Flows (Volume and Flushing Time)

Hydrology is a key parameter in determining the characteristics and health of any wetland system. Each lake and fen system is a unique entity within its complex hydrology, often deriving water from a number of sources. Within the Broads, ecological processes provide one of the fundamental drivers for the storage, cycling or export of fresh and salt water, sediment, carbon and nutrients.

Water volume and the rate at which water moves through the system (rivers and broads and fens) has a significant effect on the biogeochemical composition of the water column and hence the water ecology (plants and animals). Water volume is determined by rainfall, evaporative loss to the atmosphere, the tides (saline incursion) and water abstraction by man.

Minimum acceptable flows to minimise damaging effects on the wetland ecosystem's functioning and functions have not (yet) been set, but are needed to sustain the ecology of (sections of) rivers. Summer flows are particularly critical in certain river stretches, for example the middle Bure where many of the broads are located. Adequate flow levels are needed to reduce the build-up of algae in the summer, dilute effluent, mitigate against saline intrusion, or maintain the short and long term water storage capacity of marshes and fens[1]. Changing the hydrology (water abstraction, draining, changing water tables) near fens or marshland may have significant irreversible impacts.

Drainage of fen and surrounding land can upset the local water balance leading to desiccation or alteration of sensitive soil-pore water chemistry. Old inventories and notes, together with more recent paleolimnology (Osbourne and Moss, 1977; Manson, 1987), have defined targets for the restoration of the early biological communities of wet fen and open waters. However, fen habitat cannot usually be restored simply by reflooding, especially where saline intrusion has occurred, as the structure of peat may change irreversibly when it is drained. On drying, peat shrinks to nearly half its volume and, when subsequently aerated, degrades very quickly. If rewetted, the peaty soils of much of Broadland have only small capacity to neutralise a rise in acidity, a rise which is particularly marked if peats have been flooded by brackish water (see section on ions below.)

Critical biogeochemical processes

1) Nutrient Budgets

Increasing levels of nutrients (particularly Nitrogen and Phosphorus) has led to eutrophication. Nutrient enrichment causes microscopic algae or phytoplankton to grow at such a rate that they become dominant. The algae make water cloudy so that larger aquatic plants, which need sunlight, struggle to survive. The dominance of algae causes a further chain reaction. Without aquatic plants the larger aquatic invertebrates such as watersnails and insects do not occur, which in turn means there is less food for fish. Three phases of water quality have been identified: (1) situation which existed in the Broads before 1900; clear water with a carpet of low-growing species; (2) situation which occurred in the middle of the 20th century of clear water, but with higher nutrient levels causing luxuriant growth of taller plant species; (3) high nutrient levels encouraged abundant growth of algae which led to the loss of aquatic plants. Most broads are currently in phase (3). Only a few broads are in phase (1) and (2).

Three measures of nutrient availability may be used to compare potential phytoplankton productivities. The first, the loading, is the amount of nutrient supplied per unit volume per unit time and must be considered alongside the wash-out rate which determines the extent to which the load is diluted and partly the rate at which it is removed from the lake. Second, the mean annual concentrations of phosphorus or nitrogen in all available forms combined (dissolved and particulate). Third, the maximum concentrations of these elements in available dissolved inorganic forms.

Besides light and temperature, P is the limiting nutrient factor in the Broads water system. It limits the primary production of phytoplankton. Background mean total phosphorus

[1] Dehydration of marshland may result in the collapse of peat whereby the soil structure changes irreversibly (sediments sink), impairing the marshes water retention capacity and resulting in a different plant communities.

concentration for the Broads is about 50 μg per litre. This means that the Broads never have been and never will be oligotrophic and the area's target P load for small and large broads and rivers, but also fens, is 50 μg per litre. Between 50 and 300 μg P per litre, the system can go either way, i.e. become algae dominated or not, depending on other biological boundary conditions such as plant and fish species present. Above 100 μg P per litre, the ecosystem will become algae dominated and above 300 μg P per litre the system will become unstable.

2) Ions

An important feature of ion chemistry is the pH buffering capacity. Stable pH values are essential to maintain existing plant communities. In the Broads, the major ion concentrations are determined partly by river discharge, and river flows may have changed through increased withdrawal of water for spray-irrigation, and perhaps increased evapotranspiration in the catchment, owing to more intensive land use. The composition is modified by upstream movement or percolation of sea water and, concerning pH and bicarbonate concentrations, by algal and aquatic plant photosynthesis. Despite large crops of photosynthetic algae and therefore rapid uptake of molecular carbon dioxide, pH remains within a very narrow range, with a mean just below 8.0. Maxima rarely exceed 9.0. Ion concentrations and hence pH values are

Salinity (Cl^-)in the Broads has increased. Rising sea water level as a result of global warming is one of the underlying reasons. If clays are flooded by salty, tidal water, structural calcium and magnesium are replaced by sodium and the cohesive strength of the clay is lost. River banks may then erode more easily and quickly. These relatively longer-term effects of saline flooding are compounded by shorter-term effects upon plant communities. Doarks (1990), for example, notes that the high floristic richness of water plants in Broadland's marsh drainage channels was sensitive to and impoverished by, chlorine levels exceeding 1000 mg per litre (about 5 per cent the chlorinity of seawater).

Critical biological processes

1) Biomass

Thresholds in biomass can be identified for water and fen ecology. For aquatic ecosystems, the biomass threshold would be the available amount of water plants to sustain (protect/shelter from predation from fish) zooplankton which grazes on phytoplankton. Numerically quantifying this is difficult and has not been investigated.

We now list possible Keystone, Flagship and Umbrella species

Keystone indicators

Daphnia Magna and Daphnia Pulex – these are large bodies of zooplankton with large grazing potential on green algae (phytoplankton).

Charophytes/stonewarts – these only grow under clear water and so are an indicator of pristine water condition.

Pike (at higher level in food chain) This feeds on Plankterious fish. However this is sometimes found in degraded water unlike Daphnia Magna and Charophytes.

Also rooted water plants (since these are sensitive to light they will only survive in clear water).

Some amphibians (e.g. Adder and grass snake) could be used an indicator of structure and quality since they breed their eggs in the water, though they have been understudied.

Flagship indicators

Bittern – this is a sight hunter, which feeds on young fish. With peak breeding figures this century at around 60 booming males in 1954, the bittern is now reduced to 2 breeding pairs in the Broads. Reasons may include a lack of management of reedbed and fen habitat, resulting in a scrub encroachment and drying out, and a decline in water quality, particularly affecting prey.

Umbrella indicators

Bittern

Water Vole and Otter – these are at the top of the food chain.

Annex C. Risk and Uncertainty in cost benefit analysis

Risk and uncertainty will be associated with both the physical outcomes associated with future environmental change and their economic consequences. Assessing the possible outcomes and the likelihood of perturbations to highly complex wetland ecosystems will inevitably be fraught with difficulty. However, this will be necessary if any economic valuation is to be considered. A range of possible impacts deriving from potential management actions - for instance a standard conserve/develop scenario - needs to be identified, the relevant physical effects quantified, and probabilities attached to each. A particularly important aspect relating to the uncertainty of physical effects, is the existence of thresholds beyond which disproportional and possibly irreversible effects occur. These will be important in an economic sense both due to the disproportional extent of the impact and the inability to reverse the consequences in the future.

Analysis should not assume that all current economic conditions will hold in the future. For instance, land use changes might be predicted for the future, perhaps due to imminent regulation or long-term trends. This might affect, for example, the quantity of nitrogen in run-off and thereby the value of the wetland as a nitrogen sink. Behaviour of individuals could also adapt to change in wetland functioning, for instance with farmers changing cropping patterns as a result of increased flooding, rather than forgoing land-use or yields altogether. These changes need to be incorporated into the analysis since they can influence projected benefits and hence the net present value associated with maintaining wetland functions.

Uncertainty as to the correct value for economic variables employed and future trends can be addressed by employing sensitivity analysis or scenario analysis. Sensitivity analysis gives more than one final answer using different figures for variables employed such as the rate of discount, the extent of a function being performed and shadow pricing ratios. This provides a range of estimates within which the true figure can be expected to fall, which is less bound by particular assumptions but might result in ambiguous recommendations. Scenario analysis envisions a number of future situations with varied parameters within the valuation model, allowing a comparison of results from different future scenarios. Thus an overall decision may be preferred despite a wide range of possible outcomes, although once again it may also provide more than one decision recommendation.

A useful distinction is between risk, to which meaningful probabilities of likely outcomes can be assigned, and uncertainty, where probabilities are entirely unknown. It has been suggested that the rate at which the future is discounted could be altered to incorporate a premium for risk, adjusted either upwards (Prince, 1985) and downwards (Brown, 1983). However, risk is better dealt with by attributing probabilities to possible outcomes, thereby estimating directly the **expected** value of future costs and benefits (Boadway & Bruce, 1984) or their 'certainty equivalents' (Markandya & Pearce, 1988), rather than in some arbitrary and often subjective addition to the discount rate which will attribute a strict (and unlikely) time profile to the treatment of risk.

As Costanza (1994, p.97) points out, "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 will 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 there are unknown features of it that may yet prove to be of value. This might be particularly relevant to ecosystems where the multitude of functions that are being performed have historically been unappreciated. One practical means of dealing with such complete uncertainty is to complement a cost-benefit criterion based purely upon monetary valuation, with a Safe Minimum Standards (SMS) decision rule (Ciriacy-Wantrup, 1952; Bishop, 1978; Crowards, 1996).