ECOLOGICAL-ECONOMIC ANALYSIS OF WETLANDS: SCIENCE AND SOCIAL SCIENCE INTEGRATION*

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* The present paper and a more policy oriented twin paper were started in the context of the Global Wetlands Economics Network (GWEN). This network aims to promote multidisciplinary and international communication and collaboration between natural and social scientists that are involved in research on wetlands processes and management. It will contribute to the exchange of information and ideas about research methods and practical application of ecosystem valuation techniques, systems analysis tools and evaluation methods with particular reference to wetlands management. It is also meant to encompass both basic social science research work and user group oriented policy analysis. The network has had four meetings between February 1995 and November 1997 (in the UK, The Netherlands, Italy and Sweden).

Abstract

Wetlands provide many important services to human society, but are at the same time ecologically sensitive systems. This explains why in recent years much attention has been focused on sustainable management strategies for wetlands. Both natural and social sciences can jointly contribute to an increased understanding of relevant processes and problems associated with such strategies. Starting from this observation, the present paper considers the potential integration of insights and methods from natural or social sciences to better understand the interactions between economies and wetlands. A multidisciplinary approach can contribute to the formulation of management and institutional measures to mitigate the pressures and impacts on wetlands. To this end, attention is paid to concepts and terminology such as functions and values, frameworks and theories, and methods and models. Opportunities for cooperation and intergrated research as well as alternative viewpoints and problems associated with integration will be highlighted.

1. Purpose and Motivation

Wetlands provide many important services to human society, but are at the same time ecologically sensitive systems. This explains why in recent years much attention has been directed towards the formulation and operation of sustainable management strategies for wetlands[1]. Both natural and social sciences can contribute to an increased understanding of relevant processes and problems associated with such strategies. This article examines the potential for systematic and formalized multidisciplinary research on wetlands. Such potential lies in the integration of insights, methods and data drawn from natural and social sciences, as highlighted in previous integrated modelling/assessment surveys (Bingham et al. 1995). Concepts and terminology such as functions and values, frameworks and theories, and methods and models are distinguished and characterised. The state of the art as presented here is mainly scientific and method oriented. It results from a combined effort of natural and social scientists, notably ecologists, hydrologists and economists. The basic aim of this interdisciplinary and international collaborative effort is to explore operational methods for ecosystem valuation, systems analysis and evaluation, with particular reference to wetlands management. More specifically, a core objective is the development of a generic assessment methodology which covers a range of temperate wetland types.

It is clear that wetlands possess significant economic value and that globally they are under heavy pressure. The immediate causes of wetland loss and degradation include over-use, land conversion and degradation, pollution, climate change and species introduction. Underlying causes are, among others, price distortions, income distribution inequalities, absence of full cost accounting, policy failures, market failures (missing prices), lack of property rights and population/urbanisation growth and consequent encroachment.

Despite the increasing recognition of the need to conserve wetlands, losses have continued. The main reason is that wetlands, throughout the world have traditionally been considered by many to be of little or no value, or even at times to be of negative value. This lack of awareness of the value of conserved wetlands and their subsequent low prioritisation by the decision-making process has resulted in the destruction or substantial modification of wetlands at an unrecognised social cost. The crux of the wetland conservation issue is non-sustainable exploitation or complete conversion fuelled by this undervaluation of wetland resource benefits. Further, a strategy based solely on formal conservation designation will not be sufficient to reverse this trend. Sustainable utilisation of wetland resources must also play its part in any necessary and sufficient policy. In this article we take function and service based values and sustainable use as starting points for a discussion of multi-disciplinary research methods.

Any integrated wetland research methodology has somehow to make compatible the very different perceptions of what exactly is a wetland system, as seen from a range of disciplinary viewpoints (Maltby et al., 1994; 1996). We review the main characteristics of wetland processes and systems in a cross-disciplinary way in succeeding sections of this paper.

2. Combining Economics and Wetlands Science

Comprehensive assessment of wetlands requires the analyst to undertake the following actions:

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[1] Wetlands are the only single group of ecosystems to have their own international convention. The call for wetland protection gained momentum in the 1960s, primarily because of their importance as habitat for migratory species. A series of conferences and technical meetings culminated in the 'Convention on Wetlands of International Importance Especially as Waterfowl Habitat' (better known as the Ramsar Convention) which came into force in 1975. In 1985 there were 38 signatories, in 1991 this increased to 60 and by 1993 the total number was 75 countries (Maltby, 1986; Dugan, 1993). Currently 600 wetland sites are listed under the Ramsar Convention, covering over 30 million hectares (<1 % of the worlds land surface).

- to determine the causes of wetland degradation/loss, in order to improve understanding of socio-economic impacts on wetland processes and attributes,
- assess the range and degree of wetland functioning especially in terms of hydrological-ecological relationships, biodiversity, and the consequences of wetland quality decline alterations and/or loss,
- assess the human welfare significance of such wetland changes, via a determination of the changes in the composition of the wetlands provision of goods and services and consequent impacts on the well-being of humans who derive use or non-use benefits from such a provision,
- assess the sustainability of wetland uses and negative impacts on the wetland caused by off-site human activities,
- carry out spatial and temporal systems analysis (via a range of methods and techniques) of alternative wetland change scenarios,
- assess alternative wetland conversion/development and conservation management strategies,
- present resource managers and policy makers with the relevant policy response options.

The physical assessment of the functions performed by a wetland is an essential prerequisite to any evaluation of a wetland's worth to society, but simply identifying these functions is insufficient. Where a wetland is under pressure from human activity which provides measurable economic benefits to society, it will be necessary to illustrate the economic value of the functions performed by the conserved wetland. The provision of such economic information is essential if an efficient level of wetland resource conservation, restoration or re-creation is to be determined.

3. Definition and classification of wetlands

There is little agreement among scientists on what constitutes a wetland. Gleick (1993) observes that even among the countries of the OECD, different countries use different definitions of "wetlands". Defining wetlands is fraught with controversy and problems, partly because of their highly dynamic character, and partly because of difficulties in defining their boundaries with any precision. Where, for example, does a wetland end and a deep water aquatic habitat start? For how long and how intensely does an area have to be flooded, or in any other way saturated with water, for it to be a wetland rather than a terrestrial ecosystem? There are no universally accepted or scientifically precise answers to these questions. Certain features, nonetheless, are clear. It is the predominance of water for some significant period of time which characterises and underlies the development of wetlands. Dugan (1990) notes that there are more than 50 definitions in current use. The Ramsar definition recommended by IUCN and other international agencies involved in wetland conservation and management is adopted here as a working definition*: 'areas of marsh, fen, peatland or water, whether natural or artificial, permanent or temporary, with water that is static or flowing, fresh, brackish or salt including areas of marine water, the depth of which at low tide does not exceed 6m'*. While lacking scientific exactness this definition conveys much of the essential character of wetlands, as well as implying the complexity involved. What is does not provide, however, is any guidance on the generic characteristics of wetlands which influence how they actually function.

Similarly, there is no universally agreed classification of wetland types, and these vary greatly in both form and nomenclature between regions. The problem of classifying such a broad resource (as provided for by the definition) is not surprising. Wetlands comprise a complex range of ecosystems. They are composed of highly contrasting but interlinked habitat types such as occurs across the hydrological gradients of river flood plains or lake margins. A wide range of classifications, have been developed The generic system produced by Cowardin et. al. (1979) for the United States Department of the Interior, is based on such factors as: salinity and pH; the characteristic vegetation and dominant plant species; the frequency and duration of flooding; and the organic or mineral composition of soils. The degree of water permanence is a dominant feature of wetlands which determines the nature of soil development and the types of plant and animal communities living at the soil surface. The first level of division within Cowardin's classification divides wetlands into 5 systems: marine, estuarine, riverine, lacustrine and palustrine; plus some 120 subsequent wetland subclasses. The Cowardin et al. (1979) approach has provided the basis for simplified methodologies (e.g. Larson et al., 1989) and has been adapted for use by the Ramsar Convention (Scott, 1989). A fundamental requirement, however, remains the need for a functional classification of wetlands and considerable scientific effort is now being directed towards that goal (Simpson et al., 1998).

4. Economics and environmental change in wetlands

Conventional economics views the fundamental, relative scarcity of resources as the underlying cause of the loss of so many wetlands, whose environmental space has been converted to other uses such as agriculture, whose water has been diverted to supply other needs, whose biota has been extensively modified or harvested, or whose capacity to absorb wastes has been overburdened or by-passed. Conservation of wetlands will be associated with opportunity costs which are the benefits forgone from possible alternative uses of the wetland. On the other hand, going ahead with these alternative activities results in the opportunity costs of foregone benefits that would otherwise be derived from the conserved wetland. Quantifying and evaluating the conservation benefits in a way that makes them comparable with the returns derived from alternative uses can facilitate improved social decision making in wetland protection versus development conflict situations. Cost-benefit analysis based on the economic efficiency criterion offers one method to aid decision makers in this context. Sustainability concerns can be introduced as a series of constraints on the cost-benefit analysis and may require the further deployment of multi-criteria decision analysis methods to aid policymakers in policy conflict and goals trade-off situations. The cost-benefit criterion may need to be modified as policy makers introduce, or respond to, concerns other than economic efficiency e.g. equity concerns, employment concerns, and zero-net loss biodiversity conservation concerns. Further, governments have now formally adopted the sustainable development policy objective, as well as imposing a range of national conservation measures and designations, complementing the Ramsar Convention, to protect wetlands.

If wetlands perform many functions and are potentially so valuable, a reasonable question would be why have these values so often been ignored and wetland losses and/or degradation allowed to continue? To some degree, the desirability of the flat, fertile and easily accessible land upon which wetlands are often found, has inevitably put some of them under pressure from other uses such as agriculture, industry and urbanisation. Some past conversion might well have been in society's best interests, where the returns from the competing land use are high. However, wetlands have frequently been lost to activities resulting in only limited benefits or, on occasion, even costs to society. (Bowers, 1983; Turner, Dent and Hey, 1983; Batie and Mabbs-Zeno, 1985). This is the result of what Turner and Jones (1991) refer to as interrelated market and intervention failures, which derive from a fundamental failure of information, or lack of understanding of the multitude of values that may be associated with wetlands. Many human activities result in external effects, such as pollution from industry or agriculture, that may have an adverse impact on sites elsewhere but for which, due to a lack of enforceable rights, no compensation is paid to those affected. Pollution of wetlands, often regarded as natural sinks for waste, has been an important factor in their degradation. Many wetlands and essential features such as their ability to supply water have traditionally been treated as public goods and exposed to 'open access' pressures, with a lack of enforceable property rights allowing unrestricted depletion of the resource. Furthermore, even where wetlands are privately owned, many of the functions they perform provide benefits off-site which the resource owner is unable to appropriate. The lack of a market for these off-site wetland functions limits the incentive to maintain the wetland, since the private benefits derived by the owner do not reflect the full benefits to society.

Conventional economics is good at comparative equilibrium analysis of systems dominated by market processes, which it evaluates in terms of economic welfare changes. Wetlands can also be considered from a broader historical and coevolutionary perspective, one that recognises the significance of locally stable 'lock-in' effects caused by non-market institutions like state-governed systems, common property or even open access features (Ehrlich and Raven, 1964; Norgaard, 1974; Common and

Perrings, 1992; Gowdy, 1994). These may create many barriers and internal constraints in social-economic systems, so that conventional assumptions about individual behaviour and market mechanisms may not be appropriate (e.g. reference dependent preference effects, Bateman et al. 1997). Welfare economics is then insufficient as the sole tool for evaluating systems change, and traditional instruments cannot be trusted to realize social welfare improvements. Economic systems utilising wetland resources at an unsustainable level may be 'locked' into such a development pattern.

5. Economics and sustainable development as a wetland policy objective

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 current activities do not impose an excessive cost and loss of options burden on future generations. It has been suggested that it is 'large-scale complex functioning ecologies' that ought to form part of the intergenerational transfer of resources (Cumberland, 1991). Since wetlands are complex multi-functional systems they are therefore likely to be most beneficial if conserved as integrated ecosystems (within a catchment) rather than in terms of their individual component parts. Sustainability implies a wider and more explicity long-term context and goal than environmental quality enhancement.

Strong sustainability can be interpreted as requiring that natural resources be considered as essential inputs in economic production, consumption or welfare; or as acknowledging environmental integrity, intrinsic value and rights in nature. Especially when environmental components are unique or environmental processes are irreversible (over relevant time horizons) the latter issues may become important. Very strong sustainability would imply that every component or subsystem of the natural environment should, in principle, be preserved. A somewhat weaker version would focus on ecosystems and environmental assets that are critical in the sense of providing life support services (such as climate control, ozone layer and topsoil provision etc. (Ayres 1993) or non-use values. An even weaker version is that only a minimum amount of certain environmental assets should be maintained because the power of technological change is such that asset substitution will be the rule rather than the exception (Turner 1993a). Environmental sustainability depends mainly on ecosytem stability, resilience and biotic diversity. Traditional welfare economics focuses more on static equilibrium than on fluctuations and cycles. As a result, it is unable to deal with stability and uncertainty in a way consistent with ecological theory. Integrated systems models and co-evolutionary models may be a step towards a more dynamic and historical understanding of change in interrelated wetland and social-economic systems.

On the basis of a strong sustainability criterion, projects should be appraised on a full life cycle basis, since most development projects impinge to some degree on the environment (Pearce, Markandya & Barbier, 1989). Sustainability constraints can be imposed upon an otherwise market-oriented, cost-benefit decision making process. By introducing physical constraints to development options, opportunities for future well-being can be preserved rather than trying to impose a structure on future utilities which may be difficult to predict and to control. Wetlands mitigation policy in the US (Marsh *et al.*, 1996) can be considered a specific form of the 'strong sustainability' strategy. The policy requires that the loss of a wetland be compensated for with an alternative wetland of equal physical quality. Of course, there are many problems associated with this scheme, such as defining a measure of physical quality of different wetlands (McCrain, 1992) and issues of locality and broader landscape interactions. Furthermore, such a sustainability orientation assumes a level of analysis and governance in which trade-offs between distinct wetland systems is feasible. Nevertheless, this scheme does illustrate how sustainability constraints might be introduced, albeit in a pragmatic way.

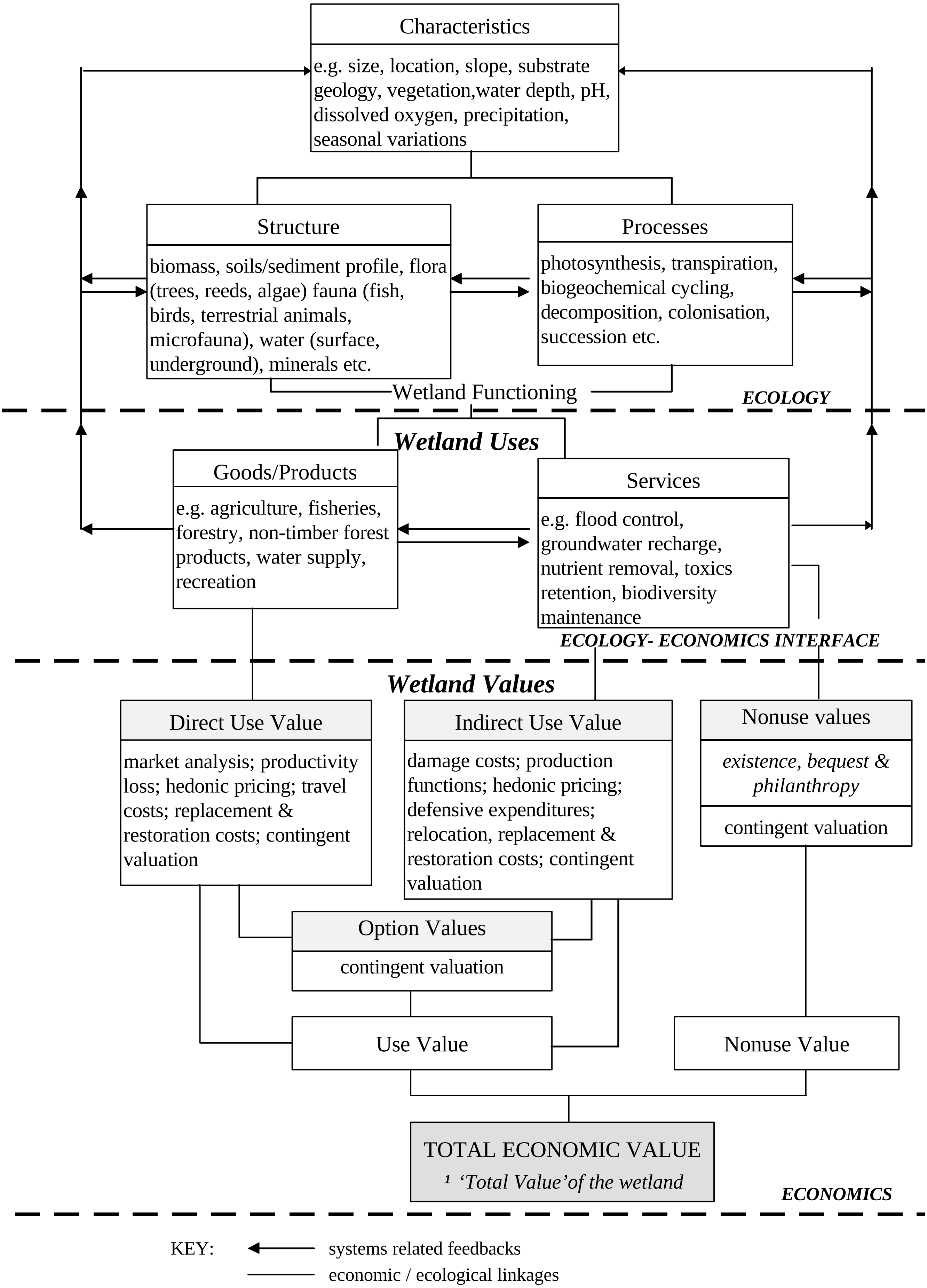
The process of environmental change manifests itself at a variety of spatial (and temporal) scales - global, regional and local. The importance of the spatial element arises from a reciprocal relationship: (1) local processes have global impacts; and (2) global trends give rise to local effects. For example, the loss of ecosystems in some regions may have a large impact on global climatological conditions and geochemical cycles. The loss/modification of peatlands affects carbon sink functioning with global warming consequences (Maltby, 1997). Over-grazing and deforestation may lead to large-scale soil erosion, downstream sedimentation, flooding and salinisation. The specific spatial environmental and economic structure surrounding wetlands will determine the sensitivity of a region to external environmental and economic forces. From a natural science perspective this will seem to be a straightforward problem of spatial demarcation. However, from a social-economic perspective defining sustainable development at a spatial or regional level is difficult. A minimum set of conditions for ‘local’ sustainability would be: (1) it should ensure an acceptable level of welfare for the regional population, which can be sustained in the future; and (2) it should not be in fundamental conflict with sustainable development at a supra-regional level. Both conditions have implications for choices about changes in wetland areas.

6. Core concepts for integrated wetlands research

A general multi-level model is particularly relevant for wetlands that provide a variety of services. The foundations of the model are provided by natural science which defines and characterises the ecosystem processes and functions. The next level contains the interrelated uses (activities) that socio-economic systems derive value from directly and indirectly; non-use values are also part of the wetland services provision. Finally, the methods and techniques available for the social valuation of wetland goods and services form the last level of the model. The links between wetland functions, services and values are illustrated in Figure 1. What is clear is that wetland uses, or the output of physical products or services, form the essential link between wetland ecology or functioning and wetland economics or values. Nonuse values will be independent of use, although they will be dependent upon the essential structure of the wetland and functions it performs such as biodiversity maintenance. Whatever the typology adopted to describe types of economic value, it will always be contingent upon the wetland 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.

The underlying objective of this approach is the development of a common way of thinking about the problems facing wetlands. This can be considered the first phase of the required research. A second phase involves case study research at selected sites, where cross-referencing is desirable. The case study research will produce as comprehensive an assessment of the ‘total’ (direct and indirect use and non-use) value of wetland processes, functions and services as is feasible; and should do this for the main wetland types. A third phase represents the final stage of the research programme in which a range of policy relevant questions should be addressed, focused on the appropriation of wetland values and possible capture mechanisms and institutional arrangements.

Figure 1: Wetland Functions, Uses and Values



7. Functional relationships within and between natural systems

Given the general typology for the assessment of wetland benefits provided in Figure 1, the first step is to recognise the immense diversity of wetlands which arise from different combinations of their many characteristics. Characteristics are those properties which describe the area in the simplest and most objective possible terms. Examples of characteristics include the biological, chemical and physical features which would describe a wetland such as, for example, vegetation, species present, substrate properties, hydrology, size and shape see Table 1. Adamus and Stockwell (1983) give 75 characteristics of wetlands, but in one sense this list is endless and site-specific. Characteristics, singly or in combination, ultimately give rise to benefits, both potential and currently realised.

From an anthropocentric viewpoint all ecosystems can be classified in terms of their structural and functional aspects (Westman, 1985; Turner, 1988; Barbier, 1989). Ecosystem structure is defined as the tangible items such as plants, animals, soil, air and water of which it is composed. By contrast, ecosystem processes refer to the dynamics of transformation of matter or energy. The interactions between wetland hydrology and geomorphology, saturated soil and vegetation more or less determine the general characteristics and the significance of the processes that occur in any given wetland. 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. They also enable the development and maintenance of the ecosystem's structure which in turn is key to the continuing provision of goods and services. Ecosystem functions are the result of interactions between structure and processes. They include such actions as flood water control, nutrient retention and food web support (Maltby et al., 1996).

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, nutrient and sediment retention, gas exchange, and pollution absorption, 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 and consequent functioning 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. An important advance is the development of an expert system-type approach whcih enables the prediction of wetland functioning on the basis of easily identifiable characteristics ('controlling variables') which can be observed in the wetland (Maltby et al., 1994; 1996).

8. Actual and potential use and non-use value relationships between humans and natural systems

It is 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. For instance, the value of flood control is likely to be limited to communities downstream of the wetland, while the value of biodiversity maintenance for recreational purposes might be spread over a wide area. The relevant population for attributing nonuse values, unconstrained by geographical distance, can be up to the global scale.

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

The significant question here is: "can all the benefits from all the classes of wetlands be classified as goods, products and services (use and non-use values)?" From Table 2 it is evident that there are strong linkages

TABLE 1: EXAMPLES OF WETLAND CHARACTERISTICS

Size
Shape
Species present
Abundance of species
Vegetation structure
Extent of vegetation
Patterns of vegetation distribution
Soils
Geology
Geomorphology
Processes (biological, chemical and physical)
Nature and location of water entry and water exit
Climate
Location in respect of human settlement and activities
Location in respect of other elements in the environment
Water flow/turnover rates
Water depth
Water quality
Altitude
Slope
Fertility
Nutrient cycles
Biomass production/export
Habitat type
Area of habitat
Drainage pattern
Area of open water
Recent evidence of human usage
Historic or prehistoric evidence of human usage
pH
Dissolved oxygen
Suspended solids
Evaporation/precipitation balance
Tidal range/regime
Characteristics of the catchment

Source: Claridge (1991)

TABLE 2: CLASSIFICATION OF BENEFITS OF WETLANDS AS SERVICES AND GOODS

SERVICES:	flood control prevention saline intrusion storm protection/windbreak sediment removal toxicant removal nutrient removal groundwater recharge groundwater discharge erosion control wildlife habitat fish habitat toxicant export shoreline stabilisation micro-climate stabilisation macro-climate stabilisation biological diversity provision cultural value provision historic value provision aesthetic value provision wilderness value provision scientific research
GOODS	forest resources agriculture resources wildlife resources forage resources fisheries mineral resources water transport water supply recreation/tourism aquaculture research site education site fertiliser production energy production

between the types of benefits. For example, the sound functioning of the 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 wetland site using a standard classification of benefits, as listed in Table 2, is an essential step before the overall value of the wetland can be derived.

Nonuse value is associated with benefits derived simply from the knowledge that a resource, such as an individual species or an entire wetland, 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 nonuse value to it. Nonuse value is closely linked to ethical concerns, often being associated with altruistic preferences, although for some analysts it stems ultimately from self-interest (Crowards, 1997 b). It can be split into three basic components, which 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, and philanthropic value is associated with the satisfaction derived from ensuring resources are available to contemporaries of the current generation. Some environmentalists support a pure intrinsic value of nature concept, which is totally divorced from anthropocentric values. Acceptance of this leads to rights and interests - based arguments on behalf of non-human nature.

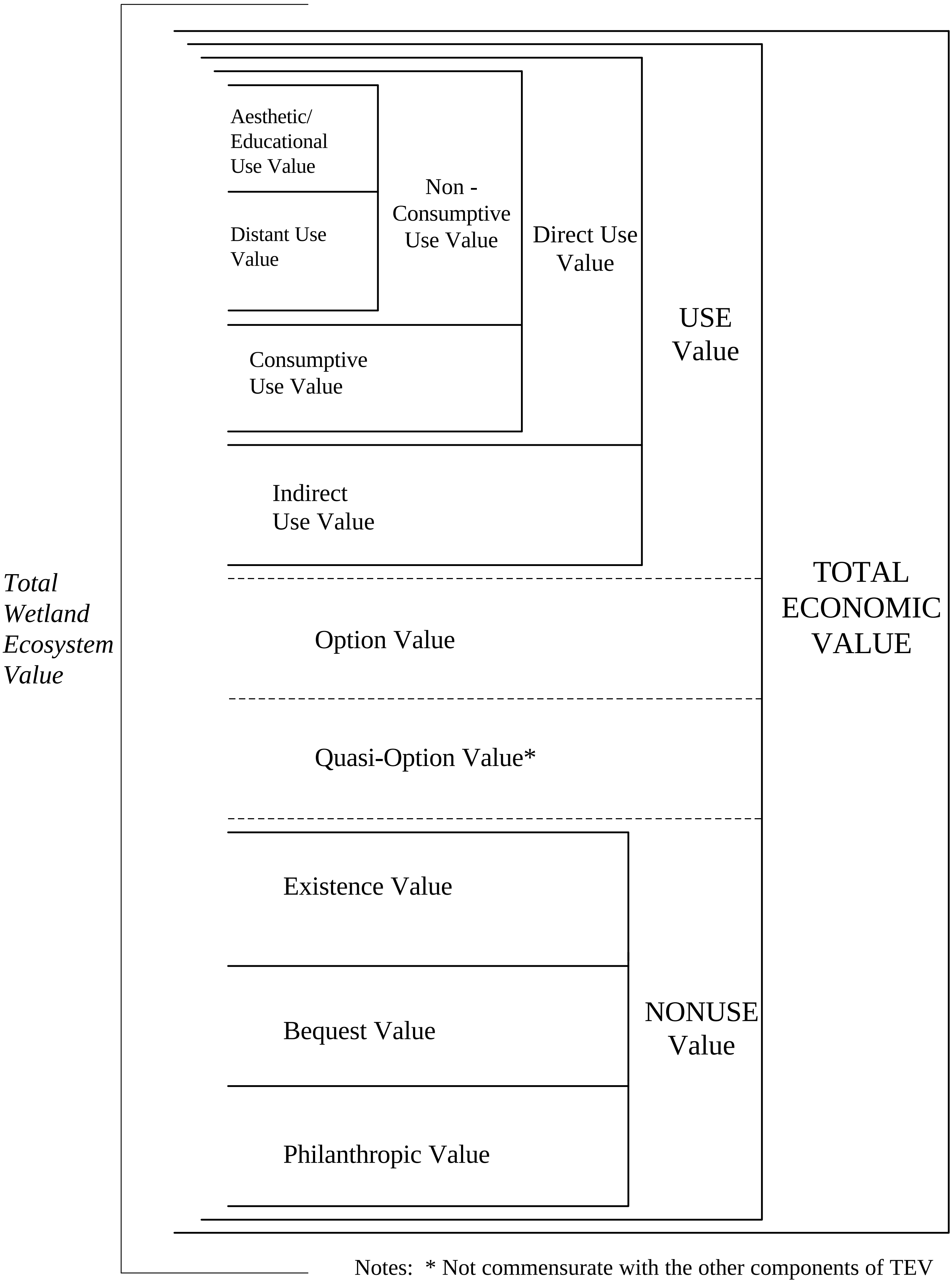
To allow for a sound judgement of wetland values there is a need to express the benefits of wetlands in a standardised fashion that is valid, repeatable, and which takes into account all possible benefits that might derive from a wetland area. Claridge (1991) points out that until now this goal seems to have been hindered by two major problems. The first is that many wetland assessments deal only with a subset of benefits which an area possesses, or else they are specific to a particular wetland type or geographical area. Too often, wetlands are evaluated by ‘experts’ (biologists, hydrologists or engineers) within the limits of their specialisation and experience, with the result that only a small sample of the range of values is described. However, if wetland evaluation is carried out against a checklist of possible benefits, and with an appropriate methodology used for expressing the value of each benefit, it will be possible to make a much more complete assessment of values. At least as important as having a complete identification of wetland benefits is the fact that such an approach will highlight the deficiencies in knowledge in relation to a particular benefit, so that action can be directed towards obtaining the missing information, or at least acknowledging such deficiencies.

9. Total economic and ecosystem value

In instrumentally valuing a resource such as a wetland, the Total Economic Value (TEV) can be broken down into a number of categories. These are illustrated in Figure 2. The initial distinction is between use value and nonuse value.

Another category is that of 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. 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 (Freeman, 1993). 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. Quasi-option value cannot be added into the TEV calculation without some double counting, it is best regarded as another dimension of ecosystem value.

Fig 2. Total Environmental Value and Total Economic Value



Notes: * Not commensurate with the other components of TEV

10. Functional and ecohydrological based wetland assessment methodology

System analysis is particularly relevant for the study of the ecological sustainability of multiple use wetlands. An important sustainability element is maintenance of biological diversity, which has three dimensions, each of which implies specific wetland features. The justification of wetland conservation policy originally focused on migrating birds, in line with the Ramsar Convention. A network of undisturbed wetlands was the goal, so that there is a set of “stepping stones” available for migrating birds, contributing to the maintenance of their diversity. The second dimension is the diversity of wetland development plant and animal species (the genetic pool). The presence of characteristic species in wetland ecosystems depends on the abiotic and biotic conditions provided by the wetland, indicated by a set of factors and resulting in a habitat type. The maintenance of biotic diversity thus implies the conservation of all the different types of habitats in wetlands. The third dimension of biodiversity relates to the nutrient removal function provided by a set of wetland plants.

The interrelated processes in a wetland ecosystem are essential elements in the sustainability of wetlands. If a set of species contributes to processes such as sedimentation or toxicant removal, management can be directed towards creating environmental conditions that are beneficial to these species. These in turn depend on equilibria caused by the physical, hydrological and chemical processes, which create the environmental conditions such as the nutrient levels, the water depth and the redox values.

In addition to these processes, more extensive spatial relations influence the structure and processes of wetlands. At the landscape level the surrounding areas influence the wetlands, by run-off or by groundwater. Sustainable wetland development policy will have to take into account the whole set of abiotic variables on which the vegetation and the fauna depend. Although biodiversity itself is not an operational variable in wetland management, processes and their driving factors can be manipulated via policy instruments and management actions to realise pre-emptive biodiversity goals.

The prediction of the processes and process changes in a wetland ecosystem is of utmost importance in the assessment of wetland functions. Many important functions are directly related to hydrology. Moreover, water is the transport medium for nutrients and other elements, including contaminants. Based on information and models of hydrological processes, nutrient fluxes, sedimentation and erosion, and even flooding can be quantified. The modelling chain can be continued with chemical modelling and the quantification of nutrient balances. Given these data, the likely presence of plant and animal species in the ecosystem may be predicted, as well as the consequent impacts on biodiversity of hydrological changes.

Different methods and models are available to improve the science of wetland systems (Mitsch et al., 1988; Anderson & Woessner, 1992; Jørgensen, 1986). Some are focused on a single dimension (i.e. Janse et al., 1992), while system modelling requires a multidisciplinary effort (i.e. Hopkinson et al, 1988; Van der Valk, 1989; De Swart et al, 1994). The models are analytical, numerical or statistical and describe a steady-state or dynamic change. Moreover, aerial photography and satellite imaging (FGDC, 1992) can be incorporated by way of GIS-systems to add spatial relations.

The development of methods for the practical assessment of wetland functioning has followed the increase over the last two decades in the intensity of wetland scientific research particularly in North America, where a multitude of biophysical methodologies have been produced to meet a range of operational requirements (Lonard and Chariain, 1985). Within the North American context the main purpose of wetland assessment has been to better inform decision makers of the publicly valuable wetland functions that may be lost or impaired by development projects (Larson and Mazzarese, 1994). The widely quoted methodology of Adamus and Stockwell (1983) originated under contract from the US Highway Administration in recognition of the potential impacts of road construction on wetlands. Both regulatory and policy instruments have driven the need for practical wetland assessment methodologies in North America but they have been generally exclusively biophysical in approach and until recently have lacked the validation of closely coupled scientific process studies.

Recent work in both the United States and Europe has focused on the possibilities of predicting wetland ecosystem functioning by their hydrogeomorphic characterisation. In the United States Brinson (1993) has outlined a hydrogeomorphic classification for wetlands which underpins a methodology involving comparison of the 'assessed' wetland with suitable reference sites (Brison, in press). A major European Union funded research initiative on the Functional Analysis of European Wetland Ecosystems (FAEWE) recognises the intrinsic value of the hydrogeomorphic approach. (Maltby et al., 1994). The FAEWE approach is based on the characterisation of distinctive ecosystem/landscape entities called hydrogeomorphic units HGMU's (Maltby et al., 1996).

Work at field calibration sites has shown that a wetland may comprise of a single HGMU or may be composed of a mosaic of various units. Empirical scientific research Europe - wide calibration sites including process studies and simulation modelling has been used to assess the validity and robustness of the hydrogeomorphic concept. Clear relationships already have been found to exist between individual HGMU's and specific wetland functions including nutrient removal and retention (Baker and Maltby, 1995), floodwater control (Hooijer, 1996), ecosystem maintenance (Clement et al., 1996) and food web support (Castella and Speight, 1996).

The functions addressed by the FAEWE procedures together with the processes identified as significant for their maintenance are given in Table 3. Each process is further subdivided and explained in terms of 'controlling' variables - environmental parameters essential to the process that support ecosystem functions. An assessment is made for all the processes of relevance or interest (e.g. relating to one, a range or all functions) based on an evaluation of each of the controlling variables identified. The controlling variables take the form of a 'decision tree' that allows the user to take a variable route through its branches depending on the answers to a series of questions. The final assessment of 'function' is based on evaluation of the combined 'process' assessments. In addition there is a 'rationale statement' providing an explanation of the outcome.

The clear link to definable variables influencing processes in the wetland ecosystem or wider landscape (e.g. supply of nitrate controlling denitrification and the nutrient export function of the wetland) provides a sound basis for linkage with economic evaluation and models of socio-economic dynamics. This linkage has been explored (Crowards and Turner 1996) and is a key objective in converting the procedures to an operational basis intended under the successor PROTOWET project (Maltby in press).

11. Systems dynamics and values

The adoption of a systems perspective serves to re-emphasize the obvious but fundamental point that economic systems are underpinned by ecological systems and not vice versa. There is a dynamic interdependency between economy and ecosystem. The properties of biophysical systems are part of the set of constraints which bound economic activity. The constraints set has its own internal dynamics which react to economic activity exploiting environmental assets (extraction, harvesting, waste disposal, non-consumptive users). Feedbacks then occur which influence economic and social relationships. The evolution of the economy and the evolution of the constraints set are interdependent; 'coevolution' is thus a crucial concept.

If 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 conceptualizing problems of scale in determining biodiversity policy is adopted, them the goal of sustaining biological diversity over multiple human generations can only be achieved, if biodiversity policy is operated at the landscape level. The value of individual species, then, is mainly in their contribution to a large dynamic, and significant financial expenditure may not always be justified to save ecologically marginal species. A central aim of policy should be to protect as many species as possible, but not all.

Ecosystem health or integrity (determined by properties such as stability and resilience or creativity), interpreted broadly, is useful in that it helps focus attention on the larger systems in nature and away from the special interests of individuals and groups. The full range of public and private instrumental and non-instrumental values all depend on protection of the processes that support the functioning of larger-scale ecological systems. Thus when a wetland, for example, is disturbed or degraded, we need to look at the impacts of the disturbance on the larger level of the landscape. Emphasis on a system-wide approach also serves to remind analysts that the social value of an ecosystem, may not be equivalent to the aggregate private total economic value of that same system's components, the system is more than just the aggregation of its individual parts, it possesses primary value (Gren et al., 1994; Turner, Perrings and Folke, 1997)

Table 3 Summary List of Functions addressed by the FAEWE Procedures

FUNCTION	Process(es) maintaining function
1. HYDROLOGICAL FUNCTIONS	**Water quantity functions**
1.1 Flood water detention	a. Short term storage of overbank flood water due to backwatering or velocity reduction b. Long term storage of overbank flood water due to impeded outflow c. Detention of surface runoff from surrounding slopes
1.2 Groundwater recharge	a. Infiltration of flood water into the wetland surface followed by percolation to a significant aquifer
1.3 Groundwater discharge	a. Upward seepage of groundwater through the wetland surface
1.4 Sediment retention	a. Net storage of fine sediments carried in suspension by river water during overbank flooding events b. Net storage of fine sediments carried in suspension by surface runoff from other wetland units or the contributory area
2. BIOGEOCHEMICAL FUNCTIONS	**Water quality functions**
2.1 Nutrient retention	a. Plant uptake of nutrients (N and P0 b. Storage of nutrients (N and P) in soil organic matter c. Absorption of N as ammonium d. Absorption and precipitation of P in the soil e. Retention of particulate nutrients
2.2 Nutrient export	a. Gaseous export of N b. Nutrient (N and P) export through land use management c. Export of nutrients (N and P) through physical processes
2.3 Peat accumulation	a. In situ C retention
3. ECOLOGICAL FUNCTIONS	**Habitat functions**
3.1 Ecosystem maintenance	a. Provision of overall habitat structural diversity b. Provision of microsites for: i. macro-invertebrates ii. fish iii. herpetiles iv. birds v. mammals c. Provision of plant and habitat diversity
3.2 Food web support	a. Biomass production b. Biomass import via physical processes: i. watercourses ii. overland flow iii. wind transport c. Biomass import via biological processes: d. Biomass export via physical processes i. watercourses ii. overland flow iii. wind transport e. Biomass export via biological processes:

12. Integrated modelling

The various perspectives of sustainability outlined in the previous section can each be examined by way of integrated economic-ecological models. The linking of models is restricted by the model type. If economic and ecological models fit in a (general) systems frame, then they may be blended in a single model structure, where compartments or modules may represent the original models, and certain outputs of one module serve as input for another. However, it is often not easy to link models directly. For instance, if both the economic and ecological systems are represented in the form of programming or optimisation models then several options are available: look for a new, aggregate objective; adopt a multi-objective or conflict analysis framework; or, when possible, derive multiple sets of optimality conditions and solve these simultaneously. However, when the economic and ecological systems are represented by different model types, it is difficult to suggest how they can be linked to one another. Where economic models have an optimisation or programming format and ecosystem models a descriptive format direct technical integration seems feasible, otherwise heuristic approaches are needed..

Model classification can take various forms. For instance, Costanza *et al.* (1993) distinguish between economic, ecological and integrated approaches on the basis of whether they optimise:

- generality: characterised by simple theoretical or conceptual models that aggregate, caricature and exaggerate;
- precision: characterised by statistical, short-term, partial, static or linear models with one element examined in much detail; and
- realism: characterised by causal, non-linear, dynamic-evolutionary, and complex models.

These three criteria are usually conflicting, and trade-offs become inevitable. In the case of systems analysis based on models for wetlands, precision is strived for at the natural science description level, while generality and realism is strived for in the description of the socio-economic-value level. The combination or integration of the two will imply a somewhat qualitative approach, although using a sequentially integrated formal approach. Interdisciplinary work, which may be the separating line between economic-ecological analysis and environmental economics or ecology, may involve economists or ecologists transferring elements or even theories and models from one discipline to another and transforming them for their specific purpose. This may require activities such as reduction, simplifying or summarising. For instance, we may come up with a simple dynamic model summarizing and simplifying some of the statistical and causal relationships of the spatial hydrological model and the statistical wetland vegetation model, and linking the outcomes to a simplified economic interaction and values model.

Examples of such statistical natural science models are *ICHORS* and *IMRAM*. ICHORS (Influence of Chemical and Hydrological Factors on the Response of Species) is a model supported by a dataset with 700 observations in an area. The modelling is based on multiple logistic regression and the resulting model can predict the response of selected wetland species to 24 abiotic variables (Barendregt and Wassen, 1989; Barendregt et al., 1993). The abiotic variables include the morphology of the surface water system, depth of the water, hydrology and water chemistry with concentration of major ions and nutrients. These variables describe the conditions of the environment in the field. Given conditions of a set with each of these variables, ICHORS estimates the probability of 150 wetland plant species found at any site with this description. Evaluation of the model output from different sets of conditions (scenarios) facilitates the identification of the presence of groups of species, characteristic of special types of vegetation. IMRAM (Influence of Environmental Factors on the Response of Aquatic Macrofauna) is a similar prediction model using abiotic information for aquatic macrofauna (Amesz & Barendregt, 1996).

A regional groundwater flow model of a research area (encompassing a number of relatively hydrologically homogeneous polders) can be constructed with MODFLOW (McDonald & Harbaugh, 1984). The spatial patterns used as input for the flow model are derived from a Geographic Information System (GIS), namely "ArcInfo"; the output from the model is stored in the same GIS. Given initial conditions, such as abstraction volumes, levels in the region and desired level of groundwater in specific areas, the model can predict quantities of groundwater throughout the region. From the MODFLOW-results the groundwater quality can be modelled with MT3D (Papadopulos, 1992). The surface water component together with water balances and surface water chemistry can also be modelled. In the GIS system, MODFLOW can be linked to ICHORS and IMRAM. Consequently, ecological response to variables driving the regions hydrology can be estimated.

13. Valuation and total value of ecosystems

A range of valuation techniques exist for assessing the economic value of the functions performed by wetlands, and these are detailed in Table 4. Many wetland functions result in goods and services which are not traded in markets and therefore remain un-priced. It is then necessary to assess the relative economic worth of these goods or services using 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 and Moran (1994), Randall (1987), Turner (1993b), and Turner & Adger (1996).

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

Where market prices exist for resources, these may have to be adjusted to provide social or shadow prices as explained above, but otherwise they are likely to provide a relatively simple means of assessing economic value. Approaches related to market analysis include the assessment of productivity losses that can be attributed to changes in the wetland and the incorporation of the wetland as just one input into the production function of other goods and services. Investment by public (especially government) bodies in conserving wetlands can represent a surrogate for aggregated individual willingness to pay and hence social value. These ‘public prices’ paid for resources can be used to approximate the value society places upon them, as for instance the costs of designating a wetland as a nature reserve. For a variety of reasons, these are unlikely to accurately reflect aggregated individual values, although techniques exist for attributing economic value based on such ‘collective choice’ decisions. (Pearce and Moran, 1994).

Table 4. Valuation Methodologies Relating to Wetland Functions.

Valuation Method	Description	Direct Use Values	Indirect Use Values	Nonuse 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)	Wetlands 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)	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 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	√	√	√

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

In order to estimate benefits given limited funds and in a relatively short time period, it may be possible to transfer data from other studies as a rough guide to appropriate values. This technique of 'benefits transfer' is, however, fraught with difficulties and subject to a number of caveats. Criteria for transferring benefits between sites are suggested by Boyle & Bergstrom (1992) as:

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

Three approaches to benefit transfer have been edentified directly transferring mean unit values; transferring unit values adjusted to suit the current study; and transferring of a benefit **function** from which unit values can be derived. A major drawback of the direct transfer of values is that no two situations will be identical and the criteria outlined above are unlikely to be met. Values will need to be adjusted when there are differences in socio-economic characteristics of households, differences in the availability of substitute or complementary goods or services, and differences in the policy setting and problem orientation. The transferring of benefit functions is likely to result in better approximation of appropriate values but is more involved than the other two approaches.

Problems common to all methods of benefits transfer remain the requirement for good quality studies of similar situations, the considerable potential for changes in characteristics between different time periods and the inability to value novel changes. Green *et al.* (1994) argue that the quality of a valuation analysis carried out using transferred benefits estimates will be no better than the quality of the transferred data itself, in the context of the study area to which it is applied. Garrod & Willis (1994, p.23) suggest that, for the UK at least, even careful modification of available benefits estimates would not "yield transfer estimates which were reliable and robust enough to be used with confidence in policy applications." Benefits transfer might, however, be more robust if it considers essential scientific variables at different sites, based on ecosystem characteristics and processes, as well as socio-economic variables. Thus a recent meta-analysis of wetland contingent valuation studies has shown that it is possible to identify wetland functions such as flood control, water supply, water quality provision and biodiversity maintenance with stated preferences (willingness to pay values) drawn from a range of wetland research studies in Europe, Canada and the USA (Brouwer et al. 1997).

14. Scenarios and evaluation

The development of prospective scenarios may be based either on an assessment of the existing policy plans and strategies for the future of the area, or on a broader approach, investigating possible options coloured by local history and politics. In addition, broad limits on economic development or structural change (land use) may be derived on the basis of a backtracking procedure, in which environmental quality levels are pre-emptively set. This could be done on the basis of a benchmark natural system structure, or on some notion of the sustainable functioning of the present or some improved natural system. Scenario building elements (controls) will include the following:

_ changes in economic activities, including land use changes and particular projects, e.g.: construction of a phosphorous removal plant, recovery of agricultural land for nature restoration, subdivision of agricultural land for residential development, and flooding of polders;
_ spatial characteristics of activities, in particular of agriculture, water abstraction, water purification, dephosphorisation, recreation, nature conservation and restoration, and residential development;

_ regulations etc. imposed on these activities, like application of manure to the land, restraining the volumes of abstracted water, and limiting the density of residential development;
_ one-off actions to redress environmental quality such as dredging and removal of nutrient-rich sediment from eutrophic lakes.

The future land use change scenarios can be linked to defined conditions in the abiotic factors, or to an overall ecosystem condition defined by functioning.. The program MODFLOW linked to ICHORS can translate these conditions to the likely spatial presence of wetland species. An economic model can then be used to calculate economic distributional effects. This suite of models will generate output in the form of a mix of spatial economic and ecological indicators. Evaluation procedures can then be deployed in order to rank various management regimes or scenarios on the basis of spatial ecological and economic indicators.

Back casting uses optimal conditions in abiotic factors to indicate, for example, the conditions of the most wanted types of vegetation in the area. Based on current plans for nature conservation and restoration, it is possible to work backwards to the required abiotic conditions needed to achieve this nature development. This 'back-calculation' will provide a set of constraints on the development of alternative management scenarios. As there are many degrees of freedom, different scenarios for a given set of ideal vegetation conditions will have to be examined. The ideal conditions are also subject to discussion, and may be considered from a range of perspectives, including cultural-natural history, biodiversity, landscape diversity, and alternative ecological sustainability and environmental quality perspectives. The main questions are how activities in the region can be structured to achieve the abiotic conditions, and how much flexibility exists for developing alternatives?

Analysts 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. Human behaviour could also adapt to changes in wetland functioning, for example, farmers changing their cropping patterns in response to 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.

A practical means of dealing with 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). 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. 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 wetlands given their role in supporting a variety of threatened species. However, it could equally well apply to irreversible impacts threatening wetland 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 wetland 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.

It will also be the case that economic efficiency although important will not be the only decision criterion of significance to resource managers and policy makers. A number of so-called multi-criteria decision support analysis methods (MCDA) have been developed 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 increasing popularity of MCDA can be attributed, in part, to the continued existence of intangible and incommensurable environmental effects which remain outside the conventional CBA calculus. It also meets the desire, in modern public decision analysis, to be presented with a spectrum of feasible solutions rather than one 'forced' solution. MCDA also allows to distinguish a number of core features of a system, or criteria for evaluation, that illustrate conflicts most clearly. These are often classified in terms of economic efficiency (or cost effectiveness), intra- and intergenerational equity and environmental quality and sustainability. In addition, weights can reflect the relative importance of each criterion considered in a particular decision context. The basis of the MCDA approach is a set of matrices which combine policy options or alternatives with a range of decision criteria. In all cases, MCDA methods require two types of information in the form of:

- an effect score matrix: the numerical assessment of all relevant impacts of a set of choice alternatives of each of them being measured in its own units; and
- a preference or weight vector: the numerical assessment of the relative priority attached to each of the decision criteria considered in the effect score matrix. A wide spectrum of techniques may then serve to find a relevant answer, depending on the specific nature of the information used and on the scope and content of the evaluation concerned.

The primary purpose of the evaluation technique is to reduce the diverse available information to either a set of single number scores, yielding a single "best" solution, or to produce a complete or partial ranking of alternatives following a series of pairwise comparisons. Nearly all MCDA techniques require the derivation of weights. In the context of wetlands research, multidimensional indicators will be needed.

15. Practical issues and problems: scale, aggegation and double counting

It is important to determine initially what the scale of assessment is going to be. This can be based on hydrological processes, uses, trade-offs between complexity of demarcated wetland area versus number and significance of impacts of off-site activities on the area studied. The geographical scale of assessment will be important. The relevant population for an economic assessment will depend in part on the type of function which is being valued. Direct use values will generally involve some contact with the wetland itself, although individuals may travel considerable distances in order to make use of the wetland. Indirect use values may be site-specific in terms of those who benefit, nonuse values are likely to be derived over a wide geographical range, but are likely to be subject to 'distance decay' away from the wetland site. Temporal scale in combination with the rate of discount applied will influence the present value of benefits attributed to wetland functioning. Calculating expected future costs and benefits involves estimating future demand for the wetland's functions. This will necessarily be unknown but assessing likely scenarios and applying sensitivity analysis can provide a range of possible values.

Quantifying wetland functioning will not in itself be sufficient. The essence of an overall socio-economic evaluation is to determine how society is affected by the functions a wetland might perform - the function itself is not intrinsically valuable. This is not, however, to argue against the fact that a certain configuration of ecosystem structure and processes is necessary for continuing resilience and functioning. It will therefore be necessary to assess features of anthropogenic regimes, upstreams and downstream of the wetland, and how these respond to changes in wetland functioning. Furthermore, economic analysis is not limited to areas with functional linkages to the wetland, but is generally more concerned with the economic region of influence and the range of relevant stakeholder interests and positions. This may conform somewhat to patterns of the local physical environment but is by no means determined by it.

If each output provided by the wetland 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 allocated explicitly 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 wetland. In areas which require 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, rather than conflicting, might be complementary. For instance, nutrient retention could promote biomass production and the possibilities for harvesting, thereby adding to the value of the nutrient retention function; on the other hand biodiversity might be reduced because of the excessive growth of a dominant plant and the supression of other species.

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Double counting will be particularly important with partial analysis and total valuation of a wetland, although some approximations to total valuation do not encounter this problem. Studies that attempt to value the wetland as a whole based on an aggregation of separate values will tend to include a certain number of functions although do not usually claim to cover all possible benefits associated with the wetland. Examples include Bishop, Boyle and Welsh (1987), Costanza, Farber & Maxwell (1987), de Groot (1994), Dixon (1989), Farber (1992), Haneman *et al.* (1991), Hanley & Craig (1991), Loomis *et al.* (1991), Ruitenbeek (1992), Thibodeau & Ostro (1981), Thomas *et al.* (1991), and Whitehead & Blomquist (1991). Studies by Gosselink *et al.* (1974), Farber & Costanza (1987) and Folke (1991) estimate the total value of a wetland in terms of its energy content, valued as its equivalent in terms of man-made energy. This can be taken as an upper bound to aggregate WTP for use values since it will include production of output that is of little use to society. However, it does not account for value of non-biotic resources such as water and soil, and it does not include any nonuse values. Shabman & Batie (1987) employ the cost of replacing a wetland as a maximum measure of the total value of services deriving from it. This is regarded as avoiding many valuation difficulties, but it does rest crucially on the assumption that these costs do not exceed the benefits derived from the wetland.

16. Conclusions

An important aspect of the economics-science interface is the possible 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 safe minimum standards. While it may not be possible to identify exact thresholds or the precise effects of crossing those thresholds that might exist, it will be important to acknowledge the possibility of approaching limits of tolerance within the ecosystem. This could be limited to identifying whether there may be no discernible threshold effects, discernible effects, or discernible effects which are likely to influence economic welfare, which will still substantially alter economic assessment of what are otherwise generally assumed to be marginal changes in outputs.

A major stumbling block in evaluating wetland benefits has been the lack of a common terminology. Authors use a confusing mix of terms, for example, "wetland functions and their social values" (Marble and Gross, 1984), "functional values" (Adamus and Stockwell, 1983), "population values" and "ecosystem values" (Mitsch and Gosselink, 1986), "attributes", "criteria" and "values" (Usher, 1986), "structure" and "function" (Turner, 1988) and "functions", "uses" and "attributes"

(Barbier, 1989). Too often, there seems to be confusion between the benefits/values of wetlands and the characteristics which are indicators of those benefits. For example, fertility and nutrient characteristics would be crucial in providing forestry and agriculture benefits, but in themselves do not represent benefits (in the anthropocentric sense). Perhaps this represents a failure on the part of managers of wetlands to define the benefits which are to be evaluated, with the result that researchers substitute as benefits those characteristics which are directly measurable. Foster (1978) points out that if useful evaluations are to be produced we first need to standardise the benefits to be measured. Some work is currently being carried out on converting lists of commonly used terminology and classification of functions to a standardised terminology (Maltby et. al., 1996).

Total Economic Value which encompasses these various types of value, is itself regarded as a part of the overall 'Total Wetland Ecosystem Value'. 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 coevolution, with the scale of economic activity exerting significant environmental pressure. The dynamics of the jointly determined system are characterized 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-organization 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.

Natural and social science researchers should reach agreement on:

- terminology and typology appropriate to valuation;
- the scale of effects to be analysed and possible associated thresholds;
- valuation methodologies;
- links between valuation and systems and scenario analysis;
- the transferability of information and results in both the scientific and economic realm;
- the focus of the analytical approach, whether thematic or by site;
- consideration of valuation within the prevailing political and social framework.

Wetlands provide a good testcase for integration of information and methods from natural and social sciences. At least four disciplines play a crucial role, namely ecology, biogeochemists, hydrology and economics. If scientists from these disciplines can develop some mutual understanding, we are halfway to a productive multidisciplinary approach which can provide for an emerging 'wetland science'. The real challenge now is its implementaion.

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