ELSEVIER

Ecological Economics 35 (2000) 7–23

ECOLOGICAL ECONOMICS

www.elsevier.com/locate/ecolecon

SPECIAL ISSUE

THE VALUES OF WETLANDS: LANDSCAPE AND INSTITUTIONAL PERSPECTIVES

Ecological-economic analysis of wetlands: scientific integration for management and policy

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Abstract

Wetlands all over the world have been lost or are threatened in spite of various international agreements and national policies. This is caused by: (1) the public nature of many wetlands products and services; (2) user externalities imposed on other stakeholders; and (3) policy intervention failures that are due to a lack of consistency among government policies in different areas (economics, environment, nature protection, physical planning, etc.). All three causes are related to information failures which in turn can be linked to the complexity and 'invisibility' of spatial relationships among groundwater, surface water and wetland vegetation. Integrated wetland research combining social and natural sciences can help in part to solve the information failure to achieve the required consistency across various government policies. An integrated wetland research framework suggests that a combination of economic valuation, integrated modelling, stakeholder analysis, and multi-criteria evaluation can provide complementary insights into sustainable and welfare-optimising wetland management and policy. Subsequently, each of the various

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PII: S0921-8009(00)00164-6

components of such integrated wetland research is reviewed and related to wetland management policy.

Keywords: Cost-benefit analysis; Ecological-economic wetland modelling; Valuation of wetland goods and services; Hydrology; Multi-criteria evaluation; Stakeholders

1. Introduction

Wetlands provide many important services to human society, but are at the same time ecologically sensitive and adaptive systems. This explains why in recent years much attention has been directed towards the formulation and operation of sustainable management strategies for wetlands. 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 formalised interdisciplinary 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 and assessment surveys (Bingham et al., 1995). The various components of integrated wetlands research will be reviewed here.

There is some disagreement among scientists on what constitutes a wetland, partly because of their highly dynamic character, and partly because of difficulties in defining their boundaries with any precision (Mitsch and Gosselink, 1993). For example, Dugan (1990) notes that there are more than 50 definitions in current use. Likewise, there is no universally agreed classification of wetland types. Classifications vary greatly in both form and nomenclature between regions; see Cowardin et al. (1979) for one influential classification system. Some features of wetlands, nonetheless, are clear. It is the predominance of water for some significant period of time and the qualitative and quantitative influence of the hydrological regime that characterise and underlie the development of wetlands. The Ramsar Convention definition, widely accepted by governments and NGO's world-wide, is as follows: '*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 6 m*'. While lacking scientific exactness, this definition conveys much of the essential character of wetlands, as well as implying the complexity involved. What it does not provide, however, is any guidance on the generic characteristics of wetlands that influence how wetlands actually function. Any integrated wetland research approach 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). In this article the main characteristics of wetland processes and systems are reviewed in a cross-disciplinary way.

Globally, wetlands are under heavy pressure. Despite the increasing recognition of the need to conserve wetlands, losses have continued. One main reason is that wetlands throughout the world are 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 priority in the decision-making process has resulted in the destruction or substantial modification of wetlands, causing an unrecognised social cost.

The paper is organised as follows. Section 2 discusses the causes of wetland degradation and loss. Section 3 presents a framework for ecological-economic analysis of wetlands. Section 4 gives a classification of stakeholders in the context of wetlands functions and values. Section 5 discusses the use of valuation techniques and cost-benefit analysis for wetland analysis. Section 6 considers the application of multi-criteria evaluation techniques for decision-making in the context of wetland management. Section 7 reviews the possibilities for integrated ecological-economic modelling. Section 8 links integrated wetland analysis to policy issues ranging from local to global levels. A final Section 9 provides conclusions and suggestions for further research.

2. Causes of wetland degradation and loss

Wetlands perform many functions that are potentially very valuable, also in economic terms. Reasonable questions would then be why these values have so often been ignored in the policy process, and why wetland losses and/or degradation have been allowed to continue. For example, coastal wetlands have been lost because of port expansion and urban and industrial sprawl all over the world (Pinder and Witherick, 1990). Despite a national policy to maintain the deltaic wetland at Koper (Slovenia) because of, for example, its important role for migrating birds, it is still endangered by two pressure forces: urban and industrial pollution, and the local governments' wish to expand built-up and industrial zones (Hesselink, 1996). Another example is the Spanish national park Coto Doñana which has been damaged as a consequence of changes in hydrology (Llamas, 1988). Farmers argued that they needed freshwater from the river for their intensive cultivation of strawberries. Despite the ecological significance of the national park, local and regional governments responded by 'correcting' the boundaries of the park and the river regime was adjusted in accordance with the farmers' request. Furthermore, tourist resort developers succeeded in constructing a tourist village on a site in the heart of the park. Similarly, the government of the Netherlands has in principle — through concessions — allowed some drilling for gas exploitation in the Wadden Sea on financial grounds, despite the fact that this area is an internationally important wetland for migrating birds from Scandinavia. The latter is true not least since most alternative wetland sites for these birds have already been lost through conversion by agriculture and industry. Other examples include the Aral Sea which suffers from water shortages because its river water supply is used upstream for cotton-fields, the Everglades in the USA which is under stress due to the inflow of nutrients from reclaimed areas where sugar cane is grown, Ireland, where extensive peatlands have been dug up for fuel, and in South East Asia, where mangrove forests have been converted to fish and shrimp cultivation ponds (Turner and Jones, 1991; Ruitenbeek, 1994; Tri et al., 1998).

These examples illustrate the conflicting interests of various stakeholder groups at different geographical scales. It deserves to be emphasised that 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 et al., 1983; Batie and Mabbs-Zeno, 1985). Why is that the case? A basic cause is the existence of market failures due to the public nature of several wetland goods and services. But what about the policies that have in fact been introduced to prevent wetlands from deterioration? Have there been policy intervention failures?

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. The Ramsar Convention, which came into force in 1975, is an inter-governmental conservation treaty, where a framework for international co-operation was provided for the conservation of wetland habitats to ensure their conservation and wise use. At present (November, 1999), 116 countries are Ramsar Contracting Parties, with 1005 wetland sites included in the Ramsar List of Wetlands of International Importance (see http://www.ramsar.org/index.html). These sites cover about 71.7 million hectares, which correspond to about 0.5% of the world's land surface. The focus of the convention on migrating birds was followed up in 1982 by the Bonn Convention (Convention on the Conservation of Migratory Species of Wild Animals), which was intended to promote international conservation measures for migratory wild animal species. Also typical wetland species are protected due to the Convention on Biological Diversity (Rio de Janeiro, 1992).

In Europe, the Council of Europe installed the Convention on the Conservation of European Wildlife and Natural Habitats (Bern Convention, 1982) in which species were linked with habitats. The EU added the Bird Directive (1979) for the protection of areas vital to birds. Later came the Directive governing the Conservation of Natural and Semi-natural Habitats and Wild Flora and Fauna (Habitat Directive, 1992). Related measures are, among others, national inventories of areas to be protected, a list of areas of EU interest, and under the heading of Natura 2000, a European Ecological Network of protected areas is to be set up.

On a national level many countries have installed national parks and nature reserves to preserve wetlands. Governmental and non-governmental listings of threatened species ('Red Lists') have added another measure to help protect wetland species from a changing wetland environment. Most countries have indirectly helped wetlands in their physical planning at national, regional and local government levels. National environmental policies have also constrained the process of change in wetlands by encouraging the maintenance or restoration of clear water, maintaining the original hydrology, and fighting the problems of acid rain or the fragmentation of the ecosystems.

The present set of regulations does not, however, seem to be sufficient. While the integration of wetlands protection strategies into different national policies has occurred, local economic development at the expense of wetlands is still quite common. Local people have used their right to improve their own conditions, without often considering the effects on a wider geographical scale. What we typically see here is 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. The information problem results because politicians and the general public insufficiently understand the role and functions of wetlands as well as the indirect consequences of land use, water management, agricultural pollution, air pollution and infrastructure for the quality and sustainability of wetlands. This is partly related to the complexity and 'invisibility' of spatial relationships among groundwater, surface water and wetland vegetation. Moreover, existing policies in different areas (environmental quality, nature protection, physical planning, etc.) are inconsistent or contradictory. Many human activities therefore 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.

In some cases, there is a long history of institutional arrangements among direct extensive users, such as common property regimes which made it impossible for 'everybody' to use the resource. The rules aimed to control and regulate the use of the wetland in such a way that the threat of overuse and overexploitation could be neutralised. In modern societies, the use of resources such as wetlands changed dramatically in a few decades. In Europe, for example, under the influence of the Common Agricultural Policy which 'subsidised' land conversion to arable regimes. The historic common property regime was based, however, on traditional forms of wetland use, and could not cope with new forms of use, such as the construction of harbours, expansion of tourism resorts, intensification of agriculture and fishing. This suggests that new property rights regimes, adapted to recent economic uses threatening the wetlands ecosystems, have to be introduced in order to prevent a further degeneration of wetlands. However, even in cases when there are well-defined property rights to wetlands, 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.

3. A framework for ecological-economic analysis of wetlands

Wetland *characteristics* are those properties that describe a wetland area in the simplest and most objective possible terms. They are a combination of generic and site-specific features. A general list would include the biological, chemical and physical features that describe a wetland such as, e.g. species present, substrate properties, hydrology, size and shape; for example, Adamus and Stockwell (1983) give 75 wetland characteristics. Wetland *structure* may be defined as the biotic and abiotic webs of which characteristics are elements, such as vegetation type and soil type. By contrast, wetland *processes* refer to the dynamics of transformation of matter or energy. The interactions among wetland hydrology and geomorphology, saturated soil and vegetation more or less determine the general characteristics and the significance of the processes that occur in any given wetland. These processes also enable the development and maintenance of the wetland structure which in turn is key to the continuing provision of goods and services. Ecosystem *functions* are the result of interactions among characteristics, structure and processes. They include such actions as flood water control, nutrient retention and food web support (Maltby et al., 1996).

These ecological concepts constitute the upper part of Fig. 1. They allow an ecological characterisation of wetlands. But an economic valuation of wetlands requires a complementary typology, since economic values depend on human preferences; what people perceive as the impact wetlands have on their well-being.[1] In general, the economic value, i.e. the benefits, of an increased (or a preserved) amount of a good or service is defined as what individuals are willing to forego of some other resources in order to obtain the increase (or maintain the status quo). Economic values are thus relative in the sense that they are expressed in terms of something else that is given up (the opportunity cost), and they are associated with the type of incremental changes to the status quo that public policy decisions are often about in practice.

The step from the ecological characterisation to economic valuation is the essential link between wetland ecology or functioning and wetland economics and values. We label this step as going from wetland *functioning* to wetland *uses* (see Fig. 1). Economic values 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 wetland *goods* and wetland *services* due to these functions. For example, fertility and nutrient characteristics would be crucial in providing forestry and agriculture benefits, but these characteristics do not in themselves represent benefits (in the anthropocentric sense). See Fig. 1 for examples of wetland goods and services; the latter may be recognised as providers of benefits that people gain without necessarily having to come in contact with a wetland.

While the total amount of resources that individuals would be willing to forego for an increased (or preserved) amount of a wetland service reveals the total economic value (TEV) of this increase (or preservation), different components of TEV can be identified (see Fig. 1). *Use value* arises from humans' direct or indirect utilisation of wetlands through wetland goods and wetland services, respectively. A value category usually associated with use value is that of option value, in which an individual derives benefit from ensuring that a resource will be available for use in the future. See, however, Freeman (1993) and Johansson (1993) on option value ambiguity. Another type of value often mentioned in the valuation literature is quasi-option value, which is associated with the potential benefits of awaiting improved information before giving up the option to preserve a resource for future use (Arrow and Fisher, 1974). Quasi-option value cannot be added into the TEV calculation without some double counting; it is best regarded as another dimension of

[1] A major stumbling block in valuing wetlands in economic terms has in fact 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, 1993), 'attributes', 'criteria' and 'values' (Usher, 1986), 'structure' and 'function' (Turner, 1988) and 'functions', 'uses' and 'attributes' (Barbier, 1989) (see also Maltby et al., 1996).

ecosystem value. *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. Nonuse value is thus independent of use, although it is dependent upon the essential structure of the wetland and functions it performs, such as biodiversity maintenance.

Various components of nonuse value have been suggested in the literature, including the most debated component, existence value, which 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. Some environmentalists support a pure intrinsic value of nature concept, which is totally divorced from an-

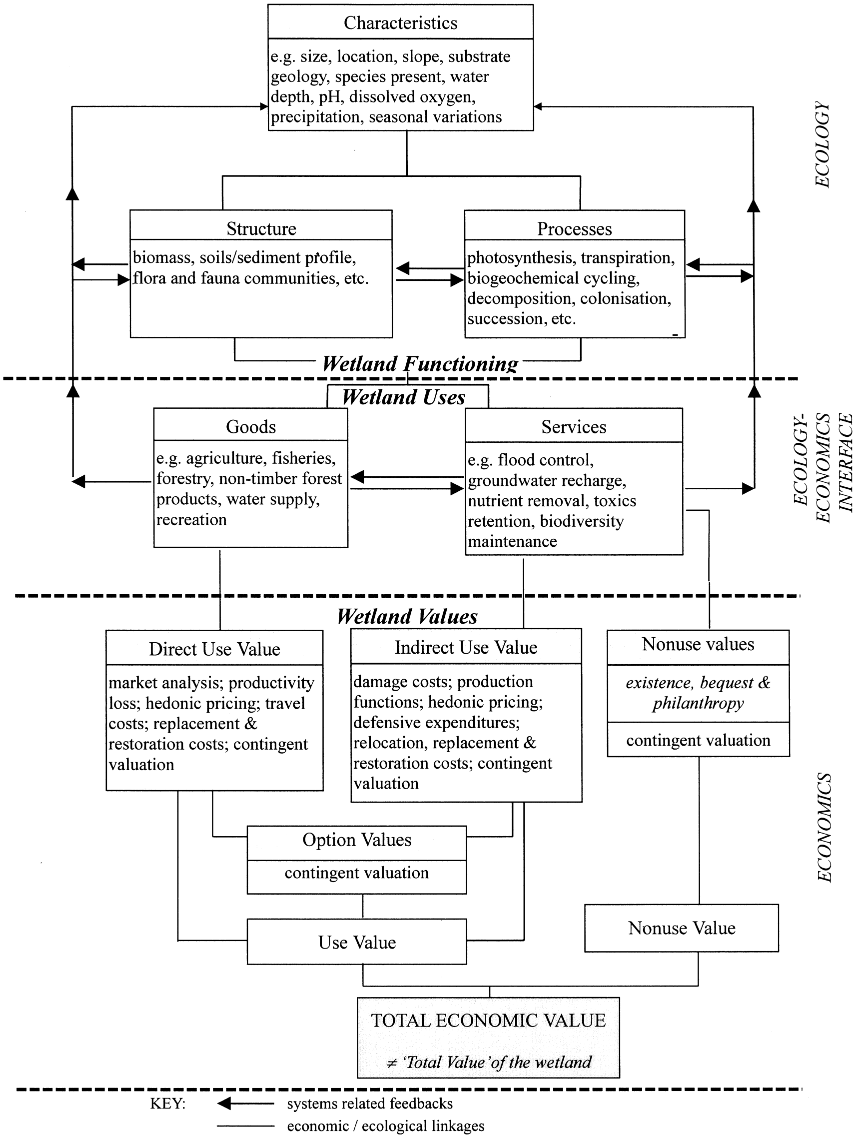

Fig. 1. Connections among wetland functions, uses and values.

thropocentric values. Acceptance of this leads to rights and interests-based arguments on behalf of non-human nature. The existence of such philosophical views is one reason why the concept of TEV should not be confused with the 'total value' of a wetland. Moreover, the social value of an ecosystem may not be equivalent to the aggregate private TEV of that same system's components; the system is likely to be more than just the aggregation of its individual parts. See Gren et al. (1994) and Turner et al. (1997) on the existence of a 'primary value' of ecosystems.

The adoption of a functional perspective is the correct way to identify wetland goods and services, but if each of them is identified separately, and then attributed to underlying functions, there is a likelihood that benefits will be double counted. Benefits might therefore have to be allocated explicitly between functions. For instance, Barbier (1994) noted that if the nutrient retention function is integral to the maintenance of biodiversity, then if both functions are valued separately and aggregated, this would double count the nutrient retention which is already 'captured' in the biodiversity value. Some functions might also be incompatible, such as water extraction and groundwater recharge, so that combining these values would overestimate the feasible benefits to be derived from the wetland. Studies that attempt to value the wetland as a whole based on an aggregation of separate values tend to include a certain number of functions although these studies do not usually claim to encompass all possible benefits associated with the wetland. Examples include Thibodeau and Ostro (1981), Bishop et al. (1987), Costanza et al. (1989), Dixon (1989), Hanemann et al. (1991), Hanley and Craig (1991), Loomis et al. (1991), Thomas et al. (1991), Whitehead and Blomquist (1991), Farber (1992), Ruitenbeek (1992), and de Groot (1994).

4. Stakeholders in wetland functions and values

Integrated ecological-economic analysis involves an identification of 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', also needs to be assessed if the full extent of economic value is to be assessed. So who are the relevant users, i.e. those who assign economic values to wetlands? It is possible to identify at least nine more or less organised groups of stakeholders:

1. *Direct extensive users* directly harvest wetland goods in a sustainable way, i.e. consistent with rapid ecosystem recovery. They thus possess a particular form of ecological knowledge enabled by an institutional setting that may be under increasing environmental change pressure. They harvest the wood for timber or fuel, the reed for roofs, the available wetland plants and fruits for produce, the waterfowl and mammals for pelts and meat, and fishes and shellfishes for food.
2. *Direct intensive users* have access to new technology that allows more intensive harvesting. In some cases there is a risk that the yield of the wetland exceeds its primary production; the wetland system loses resilience and the stocks are depleted. In other cases, such as the harvesting of biomass from fenland and subsequent fuel or feedstock production, ecosystems integrity can be maintained and enhanced.
3. *Direct exploiters* dredge the sediments in the wetland, or exploit mineral resources, peat, clay and sand without due concern for the 'health' of the wetlands.
4. *Agricultural producers* drain and convert wetlands to agricultural land, since, at least in the short to medium run, the soil is fertile, nutrients are plentiful and water is freely available.
5. *Water abstractors* use wetlands as sources of drinking water, agricultural irrigation, flow augmentation, etc. These practices may result in a wetland suffering a fall in its water table and consequent quality degradation, or in the diversion of 'polluted' water into the wetland.
6. *Human settlements close to wetlands*. Many wetlands are located in the transition zone from land to water, and may thus constitute convenient areas for the expansion of human settlements and their infrastructure; a paradox is often evident as the very presence of water is a valuable amenity that needs to be safeguarded.

7. *Indirect users* benefit from indirect wetland services such as storm abatement, flood mitigation, hydrological stabilisation and water purification to individuals and communities across large catchment areas; because of the extensive spatial provision of such services many recipients will be unaware of their origin.
8. *Nature conservation and amenity groups* combine nature conservation objectives with an enjoyment of the presence of plant and animal species. This aesthetic value of wetlands is often mixed with recreation usage values.
9. *Nonusers* may, geographical distance notwithstanding, attribute nonuse value to wetlands, possibly due to their recognition of intrinsic value in wetlands.

Clearly not all stakeholder interests are mutually compatible and the potential for value conflict is high. Policy makers are therefore required to undertake complex trade-off procedures and would benefit from the provision of integrated economic data and analysis.

5. Monetary valuation techniques and cost-benefit analysis

A range of valuation techniques exists for assessing the economic value of goods and services provided by wetlands (Fig. 1). Many wetland functions result in goods and services that are not traded in markets and therefore remain un-priced. It is then necessary to value these goods or services using non-market valuation techniques. For details on these techniques, see, e.g. Dixon and Hufschmidt (1986), Mitchell and Carson (1989), Braden and Kolstad (1991), Freeman (1993), Hanley and Spash (1993), Turner (1993b), Pearce and Moran (1994), Bromley (1995), Turner and Adger (1996), and Bateman and Willis (1999). For surveys of application of various valuation methods to wetlands, see, e.g. Gren and Söderqvist (1994), van Ierland and de Man (1996), and Barbier et al. (1997). The potential transfers of estimated wetland benefits to settings other than those originally studied, known as 'benefits transfer', is discussed in Green et al. (1994), Willis and Garrod (1995), and Brouwer et al. (1997).

Quantifying and evaluating wetland 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 (CBA) based on the economic efficiency criterion offers one method to aid decision-makers in this context. In order to be comprehensive, a CBA of a proposed policy affecting a wetland should take into account the policy's impact on the wetland's provision of goods and services. However, it should be clear from the preceding section that such predictions typically require detailed knowledge of how the policy would affect wetland functioning, i.e. the basis for the provision of goods and services. This knowledge is often imperfect and qualitative in nature. In particular, to predict in detail a policy's impact on such wetland functioning as, for example, nutrient and sediment retention, gas exchange, and pollution absorption, for any given segment of landscape, is in many cases likely to push present ecological knowledge beyond its bounds. Even wetland structure is incompletely known; for example, changes may affect the insect fauna, or soil fungi, and many of these species may never even have been described taxonomically (Westman, 1985). Adaptations of CBA to address issues of ecological complexity, notably relating to irreversibility and foregone preservation benefits, are useful in performing CBA of extreme scenarios regarding wetlands context (Krutilla and Fisher, 1975; Porter, 1982; Hanley and Craig, 1991; Hanley and Spash, 1993).

Two important conclusions follow from these observations, and they will be further discussed in subsequent sections. Firstly, in order to make CBAs of wetland policies more reliable, the economic valuation of wetland goods and services has to be as comprehensive as possible. This calls for integrated modelling of the links between wetland ecology (characteristics, structure, processes and functioning) and wetland economics (the demand for the goods and services supplied by wetlands). Secondly, even if improvements in CBAs as a basis for decision-making are desirable, it is clear that the outcome of a CBA is not on its own sufficient. The CBA criterion relies on a particular ethical basis, and it may need to be complemented as policy-makers introduce, or respond to, concerns other than economic efficiency. Moreover, the lack of detailed, quantitative knowledge of wetland functioning (in practice) precludes a full economic valuation of wetlands.

6. Multi-criteria evaluation for wetland conservation

Multi-criteria decision analysis (MCDA) offers one way 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 (Nijkamp, 1989; Janssen, 1992). MCDA typically includes multiple criteria, such as economic efficiency, equity within and between generations, environmental quality and various interpretations of sustainability. For example, various versions of 'strong' and 'weak' sustainability have been suggested in the literature, see, e.g. Pearce et al. (1989), Ayres (1993) and Turner (1993a). Weights can reflect the relative importance of each criterion considered in a particular decision context. A MCDA may thus illustrate how a particular policy would impact on and influence the various stakeholder groups introduced in Section 4.

Governments have now formally adopted sustainable development as a policy objective, as well as imposing a range of national conservation measures and designations, complementing the Ramsar Convention, to protect wetlands. Sustainability concerns can be introduced as a series of constraints on an otherwise market-oriented and CBA-based decision-making process. For example, a practical means of dealing with uncertainty is to introduce a safe minimum standard criterion (Ciriacy-Wantrup, 1952; Bishop, 1978; Crowards, 1996). By introducing physical constraints on development options, opportunities for future well-being can be preserved rather than trying to impose a structure on future preferences which may be difficult to predict and to control. 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).

Wetlands are complex multi-functional systems, and 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. In this respect, concepts such as ecosystem health or integrity (determined by properties such as stability and resilience or creativity), interpreted broadly, are 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 across the larger level of the landscape.

A strength of a MCDA is that it provides both ecological and economic information as a basis for decision-making. A separate issue is, however, to what extent this information would in fact be taken into account in real policy-making situations. Ecological information may not adequately influence the final decisions in the socio-economic system. For example, short-term commercial interests and related financial gains may appear to be more persuasive than longer-term ecological conservation arguments. The economic information provided by a CBA would perhaps be a more powerful and pragmatic support for conservation interests. But there may be a paradox here. A comprehensive CBA would rely also on a quantification of benefits due to non-market wetland goods and services, possibly also including nonuse value, if the benefit estimation involved the use of contingent valuation techniques. At the first glance, these benefits have the same configuration as 'normal' (market-based) economic information; both types of information are measured in monetary terms. On the other hand, nonuse value is hypothetical in the sense that it is not revealed by market behaviour. Such information may be a good tool to influence the perception of decision-makers and citizens regarding the high value of wetlands, but its influence in decision-making is likely to be limited by its non-market character, and the opportunities it raises for opponents to challenge its 'subjective' basis in formal proceedings or court cases.

7. Integrated ecological-economic modelling of wetlands

Integrated modelling comes in two forms. One strives towards a single model, while the other employs a system of heuristically connected submodels. Coupling wetland ecology and wetland economics within one integrated model inevitably involves compromises and simplifications. In general, in systems analysis based on models for wetlands a trade-off is needed between generality, precision and realism (cf. Costanza et al., 1993). Interdisciplinary work may involve economists or ecologists transferring elements or even theories and models from one discipline to another and transforming them for their specific purpose. For example, a simple dynamic model summarising and simplifying some of the statistical and causal relationships of a spatial hydrological model and a statistical wetland vegetation model can be linked to the outcomes to a simplified economic model. A number of approaches to integrated modelling exist, based on generalised input-output models, nonlinear dynamic systems models, optimisation models, land use models linked to geographical information systems (GIS), and mixed models. Important elements for integration are connected scenarios, models and indicators, and the arrangement of consistency among units, spatial demarcations, and spatial aggregation of information in various submodels. An overview of integrated modelling approaches and applications is given in van den Bergh (1996).

Considerable effort is devoted to increasing the precision at the natural science description level in order to facilitate the linking to the socio-economic level. The prediction of processes and process changes in a wetland — both short and long term — 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, 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 (Jørgensen, 1986; Mitsch et al., 1988; Anderson and Woessner, 1992). Some are focused on a single dimension (e.g. Janse et al., 1992), while system modelling requires a multidisciplinary effort (e.g. 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. For an example of how a statistical wetland vegetation model is linked to a regional groundwater flow model of a wetland area within a GIS framework, see Barendregt and Wassen (1989), Barendregt et al. (1993), and Amesz and Barendregt (1996).

The development of methods for the practical assessment of wetland functioning has followed the increase in the intensity of wetland scientific research over the last two decades. In particular, this has been the case in North America, where a multitude of biophysical methods has been produced to meet a range of operational requirements (Lonard and Clairain, 1995). 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 (Adamus and Stockwell, 1983; Larson and Mazzarese, 1994). Both regulatory and policy instruments have driven the need for practical wetland assessment methods 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. Efforts have also been made to establish functional classifications of wetlands (Simpson et al., 1998). Brinson (1993) has outlined a hydrogeomorphic classification for wetlands which underpins a methodology involving comparison of the 'assessed' wetland with suitable reference sites (Brinson et al., 1999).

A European research initiative (Functional Analysis of European Wetland Ecosystems, FAEWE) recognises the intrinsic value of the hydrogeomorphic approach, and is based on the characterisation of distinctive ecosystem/landscape entities called hydrogeomorphic units (HGMU) (Maltby et al., 1994, 1996). Work at field calibration sites has shown that a wetland may be comprised of a single HGMU or may be composed of a mosaic of various units. Empirical scientific research at Europe-wide calibration sites, including process studies and simulation modelling, have been used to assess the validity and robustness of the hydrogeomorphic concept. Clear relationships already have been found to exist between individual HGMUs and specific wetland functions including nutrient removal and retention (Baker and Maltby, 1995), floodwater control (Hooijer, 1996), ecosystem maintenance (Climent et al., 1996) and food web support (Castella and Speight, 1996). Links to economic valuation of fractions have also been set out (Crowards and Turner, 1996; Maltby, 1998). See van den Bergh et al. (1999) for a recent example of a wetland study for the Netherlands that employs a system of integrated hydrological, ecological and economic models. This study adopts a spatial disaggregation into 73 polders and uses a multi-criteria evaluation procedure to aggregate environmental, economic and spatial equity indicators.

8. Mitigating present failures through institutions and policies

It was illustrated above that decisions about wetlands often are characterised by inconsistencies in terms of geographical scale; local versus national versus international versus global scale. Three important ways to mitigate these inconsistencies are: (1) to create awareness of wetland values on all levels; (2) to clarify the division of responsibilities between different decision levels in order to arrive at a consistent hierarchy of decisions; and (3) to encourage local institutional arrangements that are consistent with sustainable wetland use. We discuss these ways in turn below, and it will be evident that they are complementary in nature.

The creation of a better awareness of wetland values is directly linked to the improved information that can be obtained from integrated ecological-economic models. In particular, recent advances in the development of such models and theory all seem to stress the importance of the overall system, as opposed to individual components of that system. The economy and the environment are 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 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.

In order to make progress in the important work of building integrated models, 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 its context, i.e. the prevailing political and social framework.

For some of these items, this paper has suggested a basis for agreement, which may serve as a platform for mutual understanding between scholars from different disciplines.

While scientific integration and the resulting improved information is a prerequisite for mitigating the fundamental failure of information discussed in the preceding section, more is needed for actually changing policies and stakeholder behaviour. This brings us to the two other forms of mitigation. Firstly, in order to arrive at a consistent hierarchy of decisions, the following levels and responsibilities may be defined:

- global: to define changes and appropriate policy responses at the global scale such as CO_2 fixation in organic soils to prevent global warming and sea level rise (these require international agreements by governments, e.g. Biodiversity and Climate Change Framework Conventions, etc.);
- international regions: to define changes in the sequence of wetlands (landscape ecology scale) such as the range of wetlands profitable for migrating birds, with breeding areas for reproduction, migrating areas with plenty of food and wintering areas to maintain the population (requiring measures such as regulation at the level of the Council of Europe);
- national: to maintain the national biodiversity including the defined national functions of wetland (requiring national instruments and national discussions on the economic and geographical development of designed areas);
- sub-national regions: to maintain the sequence of wetlands in a county or province (requiring regulations available on that regional level such as national park or nature reserves conservation powers);
- local: to maintain the present biodiversity and local financial returns, available from the local wetlands (requiring local regulations restricting usage, but also mandates given to regional/local authorities to balance the interests of multiple stakeholders in wetland are as surrounding catchment areas, e.g. trade-off navigation, recreation and amenity and nature conservation goals in a wetland area).

International cooperation and agreements within the first three levels would enable an international optimisation of the sequence of wetland areas. Relevant sequences of wetlands include those which would facilitate the use by migrating birds of their complete migrating routes, and an international network of wetlands which would maintain all the flora and fauna characteristic to wetlands. As many wetlands are of international significance and in this sense a global heritage, their protection should also be the responsibility of the international community buttressed by a new Global Ecological Framework to strengthen measures such as the Ramsar Convention. An extension of the Global Environment Facility, for example, could be made in order to finance wetland protection schemes.

Many important economic decisions are, however, taken at the local or regional levels, both affecting and influenced by the local economy or the functions provided by the wetlands. International and national regulations often fail to address the local subtleties involved in multiple use wetland areas. The EU's Habitats Directive, for example, has at its core a rather static interpretation of conservation. This becomes problematic for local/regional agencies which have a mandate to balance a range of stakeholder interests and to manage a rate of change in a dynamic ecological system. Local interest groups are also difficult to influence if the case being made requires an appreciation of the 'wider' benefits of wetland protection, up to the global scale of significance.

A key to resolving present failures thus seems to be behavioural change at the local level, the third form of mitigation mentioned above. 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 (Burgess et al., 2000). Likewise, special attention should be paid to existing stakeholder structure, and potentially existing local ecological knowledge and local institutional arrangements for maintaining wetlands (cf. Berkes and Folke, 1998). Such institutions may constitute a basis for building wetland management institutions that have already gained social acceptance at the local level, in contrast to governmental regulations imposed in a top-down fashion.

9. Conclusion and prospect

Wetlands all over the world are threatened in spite of various international agreements and national policies. A number of reasons have been identified here. Market failures exist due to the public good nature (a lack of enforceable property rights) of certain wetland goods and services, as well as to externalities from users (e.g. agriculture, industry, water abstraction) upon other stakeholders, including nonusers. A failure of information and lack of understanding of the multitude of values associated with wetlands is largely due to the complexity and 'invisibility' of spatial relationships between groundwater, surface water and wetland vegetation. In addition, there have been policy intervention failures, notably a lack of consistency among policies in different areas (e.g. economic, agriculture, environment, nature protection, physical planning).

Integrated wetland research combining social and natural sciences can help to partly solve the information problem and to get a grip on the required consistency among various government policies. A framework was presented that suggests that a combination of economic valuation, systems modelling, stakeholder analysis, and multicriteria evaluation can provide complementary insights into sustainable and welfare-optimising wetland management and policy. A recent European study underpins the importance of combining the various techniques to arrive at a comprehensive understanding of sustainable solutions to wetland degradation and loss (Turner et al., 1999). For example, integrated models can provide detailed information about the ecohydrological consequences, and associated economic costs and benefits, of land use policies. Valuation studies can provide insights about the loss of nonuse values associated with particular policies. Moreover, such values can be used to support economic accounting modules in integrated wetland models.

In order to make progress, further and intensified co-operation is needed between social and natural scientists. This can be done along at least two different lines. Integrated models can be developed that connect in a systematic and coherent way knowledge and theories of the various sciences. Such an approach is ambitious and requires detailed description of natural and socio-economic processes at a spatially disaggregate level. As an alternative, 'heuristic integration' can take place in which, for instance, scenarios are developed that reflect realistic changes due to (lack of) wetland management. Economic valuation experiments based on these scenarios can subsequently cause integration via individuals' assessment — through assigning values — of the relative (un)desirability of the changes implied by the scenarios. A combination of the two approaches would make it possible to refine the presentation of scenarios to individuals. This would involve a sequential process: developing an integrated model of a wetland system; using this model to inform individuals about the consequences of a proposed scenario of change; and present results via geographical information systems and computer visualisation of impacts, for instance, on the level of landscapes. Given that the research effort is multidisciplinary enough to allow due attention to the social context of the scenarios and the consequences of the institutional arrangements more or less explicitly implied by them, such a process would imply improved valuation experiments.

Acknowledgements

We are greatful to Guest Editor William J. Mitsch and six anonymous referees for useful comments on earlier versions of this paper.

References

Adamus, P.R., Stockwell, L.T., 1983. A Method for Wetland Functional Assessment. In: Critical Review and Evaluation Concepts, vol. 1. US Department of Transportation, Federal Highway Administration, Office of Research and Management, Washington, DC.

Amesz, M., Barendregt, A., 1996. IMRAM: een voorspellingsmodel voor aquatische macrofauna in Noord-Holland. Faculteit Ruimtelijke Wetenschappen, Universiteit Utrecht, Utrecht, The Netherlands, 47 pp.

Anderson, M.P., Woessner, W.W., 1992. Applied Groundwater Modeling-Simulation of Flow and Advective Transport. Academic Press, Inc, San Diego, CA 381, pp.

Arrow, K.J., Fisher, A.C., 1974. Environmental preservation, uncertainty, and irreversibility. Q. J. Econ. 88 (1), 312–319.

Ayres, R.U., 1993. Cowboys, cornucopians and long-run sustainability. Ecol. Econ. 8, 189–207.

Baker, C.J., Maltby, E., 1995. Nitrate removal by river marginal wetlands: factors affecting the provision of a suitable denitrification environment. In: Hughes, J., Heathwaite, A. (Eds.), Hydrology and Hydrochemistry of British Wetlands. John Wiley, Chichester, UK, pp. 291–313.

Barbier, E.B., 1989. The economic value of ecosystems: I-tropical wetlands. LEEC Gatekeeper Series, No. 89–01. London Environmental Economic Centre, London.

Barbier, E.B., 1994. Valuing environmental functions: tropical wetlands. Land Econom. 70 (2), 155–173.

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

Barendregt, A., Wassen, M.J., 1989. Het Hydro-Ecologisch Model ICHORS (versies 2.0 en 3.0); de Relaties Tussen Water- en Moerasplanten en Milieufactoren in Noord-Holland. Vakgroep Milieukunde, Utrecht University, The Netherlands, 72 pp.

Barendregt, A., Wassen, M.J., De Smidt, J.T., 1993. Hydroecological modelling in a polder landscape: a tool for wetland management. In: Vos, C., Opdam, P. (Eds.), Landscape Ecology of a Stressed Environment. Chapman and Hall, London, pp. 79–99.

Bateman, I.J., Willis, K.G., 1999. Valuing Environmental Preferences: Theory and Practice of the Contingent Valuation Method in the US, EU and Developing Countries. Oxford University Press, Oxford, UK, 645 pp.

Batie, S.S., Mabbs-Zeno, C.C., 1985. Opportunity costs of preserving coastal wetlands: a case study of a recreational housing development. Land Econ. 61, 1–9.

van den Bergh, J.C.J.M., 1996. Ecological Economics and Sustainable Development: Theory, Methods and Applications. Edward Elgar, Cheltenham, UK.

van den Bergh, J.C.J.M., Barendregt, A., Gilbert, A., et al., 1999. Integrated analysis of wetlands: The Dutch Vechtstreek case study. ECOWET report. Department of Spatial Economics and Institute of Environmental Studies, Free University, Amsterdam, 140 pp.

Berkes, F., Folke, C. (Eds.), 1998. Linking Social and Ecological Systems: Management Practices and Social Mechanisms for Building Resilience. Cambridge University Press, 459 pp.

Bingham, G., Bishop, R., Brody, M., et al., 1995. Issues in ecosystem valuation: improving information for decisionmaking. Ecol. Econom. 14, 73–90.

Bishop, R.C., 1978. Endangered species and uncertainty: the economics of a safe minimum standard. Am. J. Agric. Econom. 60, 10–18.

Bishop, R.C., Boyle, K.J., Welsh, M.P., 1987. Toward total economic valuation of Great Lakes fishery resources. Trans. North Am. Fish Soc. 116, 352–373.

Bowers, J.K., 1983. Cost-benefit analysis of wetland drainage. Environ. Plan. A 15, 227–235.

Braden, J.B., Kolstad, C.D. (Eds.), 1991. Measuring the Demand for Environmental Quality. North-Holland, Amsterdam.

Brinson, M.M., 1993. A hydrogeomorphic classification for wetlands. Wetlands Research Program Technical Report WRP-DE-4. US Army Corps of Engineers Waterways Experiment StationVicksburg, MS, USA.

Brinson M.M., Smith, R.D., Whigham, D.F., Lee, L.C., Rheinhardt, R.D., Nutter, W.L., 1999. Progress in development of the hydrogeomorphic approach for assessing the functioning of wetlands. In: McComb, A.J., Davis, J.A. (Eds.), Wetlands for the Future. Proceedings of INTECOL's Fifth International Wetlands Conference. Gleneagles Press, Adelaide, Australia.

Bromley, D.W., (Ed.), 1995. The Handbook of Environmental Economics. Blackwell, Oxford, 705 pp..

Brouwer, R., Langford, I.H., Bateman, I.J., Turner, R.K., 1997. A Meta-Analysis of Wetland Contingent are Valuation Studies, CSERGE Working Paper. University of East Anglia, Norwich, UK.

Burgess, J., Clark, J., Harrison, C., 2000. Knowledges in action: an actor network analysis of a wetland agri-environment scheme. Ecol. Econ. 35(1), 119–132.

Castella, E., Speight, M.C.D., 1996. Knowledge representation using fuzzy coded variables: an example with Syphidae (Insecta, Diptera) in the assessment of riverine wetlands. Ecol. Model. 85, 13–25.

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

Climent, B., Maltby, E., Hogan, D.V., McInnes, R.J., 1996. Relationships between vegetation, hydrology and soil properties in river marginal wetlands of the Torridge basin. In: Merot, P., Jigorel, A. (Eds.), Hydrologie dans les pays

Celtiques, Proc. Interceltic Coll. Hyd. Water Mgt, INSA de Rennes, France, pp. 305–314.
Costanza, R., Farber, C.S., Maxwell, J., 1989. Valuation and management of wetland ecosystems. Ecol. Econ. 1, 335–361.
Costanza, R., Wainger, L., Folke, C., Mäler, K.-G., 1993. Modelling complex ecological economic systems. Bioscience 43 (8), 545–555.
Cowardin, L.M., Carter, V., Gollet F.C., LaRoe, E.T., 1979. Classification of Wetlands and Deep Water Habitats of the United States. US Fish and Wildlife Service Publication FWS/OBS-79/31, Washington, DC.
Crowards, T.M., 1996. Addressing uncertainty in project evaluation: the costs and benefits of safe minimum standards. Global Environmental Change Working Paper GEC 96-04, Centre for Social and Economic Research on the Global Environment (CSERGE), University of East Anglia, Norwich.
Crowards, T.M., Turner, R.K., 1996. FAEWE Sub-project Report: Economic Valuation of Wetlands. CSERGE Report to the EU's FAEWE Research Project, (CSERGE). University of East Anglia and University College London.
Cumberland, J.H., 1991. Intergenerational transfers and ecological sustainability. In: Costanza, R. (Ed.), Ecological Economics: The Science and Management of Sustainability. Columbia University Press, New York.
De Swart, E.O.A.M., Van der Valk, A.G., Koehler, K.J., Barendregt, A., 1994. Experimental evaluation of realized niche models for predicting responses of plant species to a change in environmental conditions. J. Veg. Sci. 5, 541–552.
Dixon, J.A., Hufschmidt, M.M., 1986. Economic Valuation Techniques for the Environment. Johns Hopkins University Press, Baltimore.
Dixon, J.A., 1989. Valuation of Mangroves. Trop. Coastal Area Manag. Manila 4 (3), 2–6.
Dugan, P.J., 1990. Wetland Conservation: A Review of Current Issues and Required Action. IUCN, Gland, Switzerland.
Farber, S., 1992. The economic cost of residual environmental risk: a case study of Louisana. J. Environ. Manag. 36, 1–16.
FGDC, 1992. Application of satellite data for mapping and monitoring wetlands. Technical Report I. Federal Geographic Data Committee, Washington, DC.
Freeman, A.M., 1993. The Measurement of Environmental and Resource Values. Resources for the Future, Washington, DC.
Green, C., Tunstall, S., Garner, J., Ketteridge, A-M., 1994. Benefit transfer: rivers and coasts. Paper prepared for the CEGB meeting on benefit transfer, Flood Hazard Research Centre, Middlesex University, UK, H.M. Treasury, Publication No.231.
Gren, I.-M., Folke, C., Turner, R.K., Bateman, I., 1994. Primary and secondary values of wetland ecosystems. Environ. Resour. Econ. 4, 55–74.
Gren, I.-M., Söderqvist, T., 1994. Economic valuation of wetlands: a survey. Beijer Discussion Paper, series no. 54. Beijer International Institute of Ecological Economics. The Royal Swedish Academy of Sciences, Stockholm.
de Groot, R.S., 1994. Environmental functions and the economic value of natural ecosystems. In: Jansson, A.-M., Hammer, M., Folke, C., Costanza, R. (Eds.), Investing in Natural Capital: The Ecological Economics Approach to Sustainability. Island Press, Washington, DC, pp. 151–168.
Hanemann, M., Loomis, J., Kanninen, B., 1991. Statistical efficiency of double-bounded dichotomous choice contingent valuation. Am. J. Agric. Econ. 73, 1255–1263.
Hanley, N., Craig, S., 1991. Wilderness development decisions and the Krutilla-Fisher model: the case of Scotland's flow country. Ecol. Econ. 4 (2), 145–164.
Hanley, N., Spash, C.L., 1993. Cost-Benefit Analysis and the Environment. Edward Elgar, Vermont, 278 pp.
Hesselink, J.P., 1996. Sustainability War Around Skocjan Bay — Economy vs Ecology, MSc thesis, Department of Environmental Studies, Utrecht University, The Netherlands.
Hooijer, A., 1996. Floodplain hydrology: an ecologically oriented study of the Shannon Callows, Ireland, PhD thesis, Vrije Universteit Amsterdam, The Netherlands.
Hopkinson, C.S., Wetzel, R.L., Day, J.W., 1988. Simulation models of coastal wetland and estuarine systems: realization of goals. In: Mitsch, W.J., Straškraba, M., Jørgensen, S.E. (Eds.), Wetland Modelling. Development in Environmental Modelling, vol. 12. Elsevier, Amsterdam, pp. 67–97.
van Ierland, E.C., de Man, N.Y.H., 1996. Ecological engineering: first steps towards economic analysis. Ecol. Eng. 7, 351–371.
Krutilla, J.V., Fisher, A.C., 1975. The Economics of Natural Environments. Johns Hopkins University Press, Baltimore, MD, 292 pp.
Janssen, R., 1992. Multiobjective Decision Support for Environmental Management. Kluwer Academic Publishers, Dordrecht, 232 pp.
Janse, J.H., Aldenberg, T., Kramer, P.R.G., 1992. A mathematical model of the phosphorus cycle in Lake Loosdrecht and simulation of additional measures. In: Van Liere, L., Gulati, R. (Eds.), Restoration and Recovery of Shallow Eutrophic Ecosystems in the Netherlands. Kluwer Academic Press, Dordrecht, pp. 119–136.
Johansson, P.-O., 1993. Cost-Benefit Analysis of Environmental Change. Cambridge University Press, 223 pp.
Jørgensen, S.E., 1986. Fundamentals in Ecological Modelling. Elsevier, Amsterdam, 389 pp.
Larson, J.S., Mazzarese, D.B., 1994. Rapid assessment of wetlands: history and application to management. In: Mitsch, W.J. (Ed.), Global Wetlands. Elsevier Science Publishers, Amsterdam, pp. 625–636.
Llamas, M.R., 1988. Conflicts between wetland conservation and groundwater exploitation: two case histories in Spain. Environ. Geolog. Water Sci. 11, 241–251.

Lonard, R.I., Clairain, E.J., 1995. Identification of methodologies and the assessment of wetland functions and values. In: Kusler, J.A., Riexinger, P. (Eds.), Proceedings of the National Wetland Assessment Symposium. Portland, ME, pp. 66–71.

Loomis, J., Hanemann, M., Kanninen, B., Wegge, T., 1991. Willingness to pay to protect wetlands and reduce wildlife contamination from agricultural drainage. In: Dinar, A., Zilberman, D. (Eds.), The Economics and Management of Water and Drainage. Kluwer, Dordrecht, The Netherlands.

Maltby, E., Hogan, D.V., Immirzi, C.P., Tellam, J.H., van der Peijl, M.J., 1994. Building a new approach to the investigation and assessment of wetland ecosystem functioning. In: Mitsch, W.J. (Ed.), Global Wetlands: Old World and New. Elsevier, Amsterdam, pp. 637–658.

Maltby, E., Hogan, D.V., McInnes, R.J., 1996. Functional Analysis of European Wetland Ecosystems — Phase I (FAEWE). Ecosystems Research Report 18. Office for Official Publications of the European Communities, 448 pp, Luxembourg.

Maltby, E., 1998. Wetlands within catchments — some issues of scale and location in linking processes and functions. In: McComb, A.J., Davis, J.A. (Eds.), Wetlands for the Future. Gleneagles Publishing, Glen Osmond, Australia, pp. 383–391.

Marble, A.D., Gross, M., 1984. A method for assessing wetland characteristics and values. Landsc. Plan. 11, 1–17.

Mitchell, R.C., Carson, R.T., 1989. Using Surveys to Value Public Goods: The Contingent Valuation Method. Resources for the Future, Washington, DC, 463 pp.

Mitsch, W.J., Straškraba, M., Jørgensen, S.E., 1988. Wetland Modelling. Elsevier, Amsterdam, 227 pp.

Mitsch, W.J., Gosselink, J.G., 1993. Wetlands, second ed. Van Nostrand Reinhold, New York, 722 pp.

Nijkamp, P., 1989. Multi-criteria analysis: a decision support system for sustainable environmental management. In: Archibuqi, F., Nijkamp, P. (Eds.), Economy and Ecology: Towards Sustainable Development. Kluwer, Dordrecht, The Netherlands.

Pearce, D.W., Markandya, A., Barbier, E.B., 1989. Blueprint for a Green Economy. Earthscan, London, 192 pp.

Pearce, D., Moran, D., 1994. The Economic Value of Biodiversity. Earthscan, London, in association with the IUCN.

Pinder, D.A., Witherick, M.E., 1990. Port industrialization, urbanization and wetland loss. In: Williams, M. (Ed.), Wetlands: A Threatened Landscape. Basil Blackwell, Oxford, pp. 234–266.

Porter, R., 1982. The new approach to wilderness appraisal through cost-benefit analysis. J. Environ. Econ. Manag. 11, 59–80.

Ruitenbeek, H.J., 1992. Mangrove Management: An Economic Analysis of Management Options with a Focus on Bintuni Bay, Irian Jaya. Environmental Management Development in Indonesia Project (EMDI) Halifax, Canada and Jakarta, Indonesia.

Ruitenbeek, H.J., 1994. Modelling economy-ecology linkages in mangroves: economic evidence for promoting conservation in Bintuni Bay, Indonesia. Ecol. Econ. 10, 233–247.

Simpson, M., Maltby, E., Baker, C., Jones, I., McInnes, R., 1998. The development of a hydrogeomorphic classification of European Wetlands, Abstract from Society of Wetland Scientists Annual Conference, Anchorage, Alaska, June, 1998: http//www.sws.org/.

Thibodeau, F.R., Ostro, B.D., 1981. An economic analysis of wetland protection. J. Environ. Manag. 12, 19–30.

Thomas, D.H.L., Ayache, F., Hollis, T., 1991. Use values and non-use values in the conservation of Ichkeul National Park, Tunisia. Environ. Conserv. 18 (2), 119–130.

Tri, N.H., Adger, W.N., Kelly, P.M., 1998. Natural resource management in mitigating climate change impacts: mangrove restoration in Vietnam. Glob. Environ. Change 8, 49–61.

Turner, R.K., Dent, D., Hey, R.D., 1983. Valuation of the environmental impact of wetland flood protection and drainage schemes. Environ. Plan. A 15, 871–888.

Turner, R.K., 1988. Wetland conservation: economics and ethics. In: Collard, D., et al. (Eds.), Economic, Growth and Sustainable Development. Macmillan, London, pp. 121–159.

Turner, R.K., Jones, T. (Eds.), 1991. Wetlands, Market and Intervention Failures. Earthscan, London, 202 pp.

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

Turner, R.K., 1993. Sustainability: principles and practice. In: Turner, R.K. (Ed.), Sustainable Environmental Economics and Management: Principles and Practice. Belhaven Press, London, pp. 3–36.

Turner, R.K., Adger, W.N., 1996. Coastal Zone Resources Assessment Guidelines. In: LOICZ-IGBP Reports and Studies, No. 4. Land Ocean Interactions in the Coastal Zone (LOICZ), Texel, The Netherlands, 101 pp.

Turner, R.K., Perrings, C., Folke, C., 1997. Ecological economics: paradigm or perspective? In: van den Bergh, J.C.J.M., van der Straaten, J. (Eds.), Economy and Ecosystems: Analytical and Historical Approaches. Edward Elgar Publishers, Cheltenham, UK, pp. 25–49.

Turner, R.K., van den Bergh, J.C.J.M., Bateman, I.J., et al., 1999. Ecological-Economic Analysis of Wetlands: Functions, Values and Dynamics (ECOWET), Final Report, Contract No. ENV4-CT96-0273, European Commission, Brussels, CSERGE, University of East Anglia, Norwich, UK, 639 pp.

Usher, M.B. (Ed.), 1986. Wildlife Conservation Evaluation. Chapman and Hall, London.

van der Valk, A., 1989. Northern Prairie Wetlands. Iowa State University Press, Ames, IA.

Westman, R.E., 1985. Ecology, Impact Assessment and Environmental Planning. Wiley, Chichester, UK.

Whitehead, J.C., Blomquist, G.C., 1991. Measuring contingent values for wetlands: effects of information about related environmental goods. Water Resour. Res. 27 (10), 2523–2531.

Willis, K.G., Garrod, G.D., 1995. Transferability of benefit estimates. In: Willis, K.G., Corkindale, J.T. (Eds.), Environmental Valuation: New Perspectives. CAB International, Wallingford, UK, 272 pp.