The **RUBICODE** Project

Rationalising Biodiversity Conservation in Dynamic Ecosystems

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# Review of the dynamics of economic values and preferences for ecosystem goods and services

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## ABSTRACT

The present review has been conducted within the EU funded coordination action project entitled *Rationalizing Biodiversity Conservation in Dynamic Ecosystems* (RUBICODE) (www.rubicode.net). It addresses both the issue of a *rational* as well as a *dynamic* approach to biodiversity conservation by focusing on published evidence on the dynamics of economic values and preferences for ecosystem goods and services. Empirical evidence referring to the dynamics of ecosystem values is identified as both demand-driven and supply-driven value dynamics. A survey of temporal reliability tests reveals considerable differences in the time span examined (2 weeks to 20 years); it is therefore difficult to be conclusive about WTP values being stable over short to medium time periods. For longer periods, both a weak and a strong version of preference evolution are examined; here the elements of cultural transmission and evolutionary approaches make the task of modelling the dynamics of preferences rather complex. Integrated models and dynamic bioeconomic models are examined as representative approaches to supply-driven dynamics. These approaches share a role in pushing our understanding of complex systems and alerting both researchers and policy makers to the dangers of oversimplification. The reviewed models are nevertheless normative in nature in the sense that they describe how the complex socio-ecological systems should evolve over time in order to fulfil the requirements of efficiency and sustainability. Mixing of methods and pooling of data seems the only way forward. In this respect, the potential of systematic and formalised interdisciplinary research lies in the integration of insights, methods and data drawn from natural and social sciences.

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'We live by destroying utilities; by eating the food,
burning the fuel, and so on; in order to have
the prospect of continuing to live, we must
re-create today the utilities we destroy today'
Schackle 1967, p. 154

# I. Introduction

Ecosystem valuation is à la mode. The by now well-known Stern report on the cost of the impacts of climate change (Stern 2007) is harnessing the fruits of the recent wave of interest in the economic assessment of ecosystem degradation. A similar effort to estimate the economic value of biodiversity loss in Europe is under way (*Review on the Economics of Biodiversity Loss*, http://ec.europa.eu/environment/nature/call_evidence.htm). Experts and decision-makers alike seem to embrace the need for sustainability performance indicators based on the monetized costs and benefits of policies to protect the environment. Ecosystem valuation is also the terrain where a number of controversial analytical decisions in ecology and economics have to be taken and it is, therefore, an extensive and multifaceted arena of hypothesis testing and empirical validation of basic assumptions and approaches.

The present review is conducted within the EU funded coordination action project entitled *Rationalizing Biodiversity Conservation in Dynamic Ecosystems* (RUBICODE) (www.rubicode.net). It addresses both the issue of a *rational* as well as a *dynamic* approach to biodiversity conservation by focusing on published evidence on the dynamics of economic values and preferences for ecosystem goods and services. It recognizes the fact that agreement over the robustness and accuracy of economic value estimates to date has far from been achieved among its practitioners. Nevertheless, stock taking, a necessary aspect of any cooperative venture - and ecosystem valuation is a cooperative venture of more than one discipline - is one among many potential factors determining the success of the enterprise. The problem of reaching an agreement on how costs and benefits of protecting the natural environment should be quantified and communicated is a non-trivial problem since the heterogeneity of national interests, institutions and priorities is bound to lead to a diverging understanding of what constitutes an acceptable valuation scheme. On the other hand, the non-binding nature of international valuation practices and initiatives leaves practically no other option open to the scientific community; either the interested parties reach a consensus on the process and outputs of ecosystem valuation or the effectiveness of proclaimed valuation applications is seriously jeopardized.

The issue of ecosystem valuation is addressed within the organizing context of *ecosystem goods and services*, as popularized by the relevant work of the *Millennium Ecosystem Assessment* (www.milleniumassessment.org). This framework is explicitly chosen in an effort to highlight at this stage the characteristics and feasibility of a common but differentiated scheme in support of integrated ecological-economic applications. The line of reasoning relies heavily on the expanding literature concerning ways to identify, analyze and quantify ecosystem goods and services, paying special attention to the newly introduced concept of *Service Providing Units* (Luck et al. 2003). A review of valuation studies and databases of ecosystem services has been undertaken with a specific interest on dynamics aspects of preference formation such as option values, discounting, insurance values, etc.

In the course of stakeholder deliberations within RUBICODE it was soon realized that an extension to non-monetary valuation techniques would have been necessary in order to gain a better understanding of social processes and normative/moral aspects of the ecosystem valuation process. Economic valuation methodologies could then be directly contrasted with a number of qualitative and inclusionary approaches to preference elicitation with the aim of better understanding and formalizing the dynamics of preferences and values. Unfortunately, it has proven impossible to realize this task within the time and resource constraints of the present review.

The objectives of this study are:

1. To assess the suitability and desirability of dynamic value estimates for environmental decision-making;
2. To systematize existing knowledge referring to the dynamics of preferences and values in the peer review literature;
3. To assess the nature of the link between the dynamics of ecosystem change and stability/evolution of ecosystem values; and
4. To draw conclusions for future research needs.

The present review addresses itself to a broad audience of conservation scientists. Given that the root problem is to justify in quantitative terms the need to '*re-create today the (ecosystem) utilities we destroy today*', environmental economists have had recourse time and again to metaphors of values. Therefore, to provide a common understanding of the key issues that underpin the need for this study, the review firstly provides a background summary of the importance of ecosystem value estimates within the environmental policy context. Section II discusses the contextual nature of ecosystem value estimates and their uses within the various policy and decision-making contexts. Section III addresses a number of issues in the estimation of ecosystem values with special attention given to the dynamics of preferences and values. Section IV reviews the empirical evidence referring to the dynamics of ecosystem values by identifying both demand-driven as well as supply-driven value dynamics. Section V provides conclusions and recommendations from the study.

## II. Ecosystem values in a policy context

This section introduces the reader to the overall context and problem setting of the present review. It documents the necessity of linking conservation goals with an estimation of the relevant social values at stake and the multifaceted aspects that such a link may take in the realm of actual environmental decision-making. The central driving ideas of 'ecosystem service' and 'service providing unit' are then explored and linked to the issue of valuation.

### II.1 The impossibility of a value-free conservation strategy

Despite undeniable progress in the conservation of natural wealth over the past 30 years, developed and developing countries still face a series of major environmental challenges: global warming, habitat degradation and species loss, collapse of renewal resource stocks, accelerating water scarcity, land contamination, urban air pollution - an endless suite of complex issues demanding a stronger commitment, a better science and a heavier financial burden. Though 'the world has been always ending' (Langford 2002), the stakes were never before so high. A number of biophysical indicators published by international agencies document this trend while environmental degradation appears intrinsically linked with issues of human rights, national security, human health and poverty (MEA 2005, EEA 2007, UNEP 2007, FAO 2006, World Bank 2006, Daszak et al. 2000, WRI 2005).

Are our measures to protect the planet counterproductive? On the 14th December 2007, the meeting of the European Council took stock of progress in the main EU policy fields. Commenting on the Commission's first progress report on the renewed EU Sustainable Development Strategy, the Council asserts that '*Sustainable development will not be brought about by policies only: it must be taken up by society at large as a principle guiding the many choices each citizen makes every day, as well as the big political and economic decisions that have. This requires profound changes in thinking, in economic and social structures and in consumption and production patterns [.......] the main focus should therefore be on effective implementation at all levels.*' (http://ec.europa.eu/environment/eussd/index.htm).

Why is the EU Sustainable Development strategy so hard to implement? It would be too naive to assume that the European Ministers blame their own citizens for disobedience to the laws. On the contrary, what this seems to imply is that the European sustainability arsenal lacks effective incentive schemes which are able to change behavioural patterns of production and consumption in an environmental friendly manner. The same is true of USA environmental policies in a number of related fields: risks to children's health, air pollution, climate change and endangered species conservation are four policy debates where, according to Shogren (1998), serious socio-economic considerations are omitted. Implementing appropriate responses to ecosystem degradation without taking behavioural, and therefore welfare, theoretic implications into account is also apparent in another major field of USA sustainability assessment: natural resource damage assessment. Damage assessment arises when parties cause injuries to a natural asset, altering its flow of goods and services. In taking restoration and mitigation measures against ecosystem damages, federal agencies (mainly the Department of Interior and the National Oceanic and Atmospheric Administration) rely on a service-to-service approach, that is, restoration measures of damaged sites are scaled and new replacement measures are planned on a functionally equivalent habitat-to-habitat basis (known as Resource Equivalency Analysis). By restoring damaged habitats and creating new, replacement habitats, the trustee agencies aim at restoring, besides ecosystem service losses, also human welfare losses and thus '*making the public whole*' (for a general review on the subject, see REMEDE at www.envliability.eu). Logical as it may seem, such an engineering approach misses some important welfare theoretic conditions that are crucial if the biophysical restoration metric used in the damage assessment and scaling exercises is going to be equivalent to a human welfare metric. Both Flores and Thacher (2002) and Dunford et al. (2004) note that the manner in which traditional Resource Equivalency Analysis handle value, makes significant a priori assumptions[1] that complicate the linkage between biophysical and economic principles of compensation.[2]

In a recent review on European management strategies for conservation, Haslett et al. (2008) note that after a sequel of conservation targets in the past (rare species, aesthetic landscapes and habitats) societies are faced with '*the dilemma as to whether to use limited and usually inadequate human and financial resources to pursue the conservation of particular species or whether to invest in the management and protection of habitats that are of notable biological*

[1] That is, homogeneity of preferences and temporal stability of the habitat value.

[2] Boyd and Wainger (2003) show why ecosystem compensation and exchange programs in the USA require benefit analysis in order to guarantee that compensation or trades preserve the social benefits lost when ecosystems are destroyed or degraded. Zafonte and Hampton (2007) offer an indication of the magnitude of such discrepancies between biophysical and value compensation metrics and conclude that under fair general conditions they are not substantial. McConnell and Bockstael (2005) assert though that 'some of the worst mistakes in environmental valuation are made by attempting to measure the social cost of environmental degradation as the cost of remediating the damage" (p. 647).

*value remains a critical issue in practical conservation strategy*' (p. 1) Indeed, modern literature on conservation strategies abound with suggestions on how to bring into action the linkage between biophysical and economic principles mentioned in the previous paragraph. A popular trend in this respect can be discerned in the current efforts of a number of international agencies, including UNEP, to develop systems of payments for ecosystem services.[3] It would take us too far to dwell on this; for the purpose of the present review it is suffice to stress that in the realm of conservation strategies 'behaviour matters'! Human behaviour vis-à-vis conservation choice unravels the issue of values and their (positive and/or normative) role in designing effective conservation strategies. [4]

The burgeoning interest on ecosystem functions, services and values within state agencies and international organizations is a testimony to the importance attached to ecosystem valuation in modern nature conservation thinking. Increased academic interest in economic valuation in the last years is documented both by the multiplication of special issues on valuation in scientific journals, as well as by the venturing of natural scientists themselves into valuation exercises of some sort.[5] The practical interest in benefit assessment for nature's services is aptly demonstrated in numerous official documents and reports from a wide variety of sources such as IUCN, WWF, CBD, UNEP, DEFRA, MEA, etc. Noteworthy is UNEP/MAP explicit attempt to introduce *resource consciousness* in its Regional Strategic Environmental Action Plan. Since, within the wider context of Integrated Coastal Zone Management: '*raw cost information is insufficient to support investment decisions*' what is needed is an investment plan where '*benefits [..] derived from the reduction or avoidance of pollution impacts on resources of social, economic and environmental value' are demonstrated. Moreover, in order for benefits estimates to be of relevance to prospective investors, their definition should include 'the conservation of resource for their existence (or non-use) value*' (UNEP 1999, p. 67-69).

Nevertheless, fifteen years after the Convention on Biological Diversity (UN 1992), the core issue of definition of biodiversity conservation priorities at a global scale remains unsettled. One of the main contentious issues is specifically related to differing perceptions and the operational definition of the intrinsic value of biodiversity that is explicitly referred to in the CBD text. CBD includes the statement that '*ultimately, all ecosystems should be managed for the benefit of humans'*. It also includes the principle of '*benefit-sharing'*. According to these assumptions, the objectives of the management of land, water and living resources are a matter of societal choice. Almost inevitably, conflicts arose over whether any framework policy text, such as the CBD, could legitimately say that all ecosystems must be managed, and if so, whether that should always be for human benefit or, on the contrary, whether it is ever legitimate to deny the right of humans to use living resources. On the one hand, many stakeholders (countries, land owners, producers, etc.) see the alleviation of poverty as the central issue for their societies and therefore view the prime function of natural resources as a means to reduce human misery. On the other hand, other stakeholders accept legislation that

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[3] For UNEP initiative see www.uneofi.org. For a general overview see Ferraro and Kiss (2002). For a critique of the ecosystem service approach to nature conservation and a plea for focusing instead on the primacy of ethic and aesthetics see McCauley (2006) and the reactions to this in Nature 443/ 19 Oct. 2006.

[4] Heal (2000) argues that '*Valuation is neither necessary nor sufficient for conservation. We conserve much that we do not value, and do not conserve much that we value*.' (p. 29) What matters most are incentives: '*Incentives are critical for conservation; valuation is not necessary for establishing the correct incentives.'* The ecologists Myers and Reichert (1997) on the contrary assert that 'We don't protect what we don't value' (p. xix).

[5] See for example the special issues in the journals *Ecosystems* 3 (2000) and *Environmental Science and Technology* 34 (2000), as well as Pimentel et al. (1997), Humphries et al. (1995), Daily et al. (2000).

forbids any human activity in designated pristine lands and some others insist on the intrinsic value of biodiversity.

So far, several categories of values have been defined: economic, aesthetic, ethical, scientific, evolutionary and ecological. The multidimensionality of natural processes adds to the confusion about the different values of its components. The various actors may have different or even conflicting perceptions on the significance of these values because of cultural differences, difficulties in calculating benefits or placing a monetary value on living entities, products or services. At the international political arena, these conflicts are overcome through formalistic compromises such as*: 'Ecosystems should be managed for their intrinsic values and for the tangible or intangible benefits for humans, in a fair and equitable way'* (see CBD/COP V Decision V/6, 2001). Rational arguments that would strengthen the perception of the public and policy-makers about the seriousness of threats to natural resources need to be established through new methodologies to valuate and evaluate the various forms of ecosystem goods and services from the perspective of all societal actors. The eighth Conference of the Parties of the Convention on Biological Diversity requested the CBD Executive to compile "*information on methods for the valuation of biodiversity resources and functions and associated ecosystem benefits…to explore with relevant organizations options for cooperative activities that strengthen existing information systems on valuation methodologies and…. to explore options for the design and application of flexible and reliable innovative tools for assessment and valuation of biodiversity resources and functions and associated ecosystem services*" (Decision VIII/5 in: CBD 2006, pp. 301 ff).

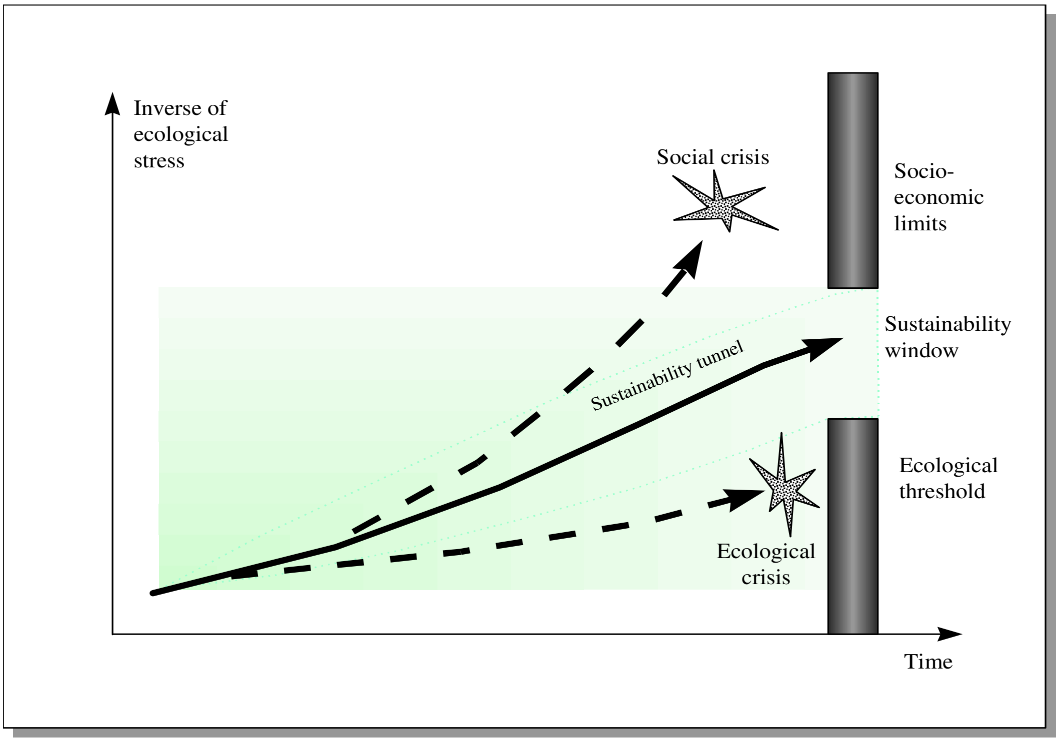
What then can scientific knowledge offer to the resolution of such conflicts, especially at local scales and within ecosystem entities that mediate multiple functions? And what can social sciences contribute? The challenge is portrayed by what J. Boyd (2007) has termed "endpoint problem': '*If linked social and natural science is a relay race, endpoints are the baton. The problem is that the baton never gets handed off smoothly*.' Boyd defines ecological endpoints as concrete statements, intuitively expressed and commonly understood, about what matters in nature. To help society make informed decisions in using space and resources we have to start by quantifying ecosystem functions and identifying needs. We can then ask what kind of knowledge and information does the policy-making process need in order to comply with a sustainable use of spaces and resources. It is evident that besides data and predictive ability regarding changes in ecological parameters, there is a fundamental need to prioritize alternative uses by means of both 'objective' ecological indicators <u>and</u> 'subjective' economic values.

## II.2 In search of policy relevant value estimates

Conservation planning has to manoeuvre itself through 'windows of sustainability' characterized by socially acceptable levels of both ecosystem integrity as well as consumption flows (Figure 1). If not, it runs the danger of failure on account of either too much conservation effort or too high consumption streams. In answering such dilemmas, political arguments take precedence over scientific and/or efficiency arguments. This is more than expected in cases where conservation targets are intertwined with equity and distributional considerations. In cases though where circumstances demand an efficiency calculus of alternative management options, human values - as revealed by actual or hypothetical choices - have a central role to play. An operational decision-support system for a better ecosystem conservation strategy and a transparent sustainability performance scheme though has still to evolve (Turner and Daily 2008).

Figure 1 portrays the rationale for using both ecological indicators and subjective values in sustainability performance. Recognizing the need for both ecological indicators and subjective values help us define the former and operationalize the latter. There are a number of ecological indicators in use for monitoring drivers, pressures and states of ecosystem goods and services (Feld et al. 2008). A number of efforts to identify and develop performance measures for evaluating the outcomes and impacts of conservation efforts on biodiversity are discussed in Tucker (2006). The author stresses the fact that increased performance measurement in the realm of ecosystem conservation is driven today by the increasing attention being given to accountability schemes within trans-national corporations, government development agencies and NGOs. Such issues have resulted in a number of studies of measures of conservation success initiated by conservation organisations.[6]

**Figure 1: Windows of sustainability.**




The increasing demand for evidence that good value for money has been achieved when public funds have been used for conservation projects puts the issue of reliable monetization of conservation costs and benefits at the forefront. Although ambitious, this kind of reporting on ecosystem health will become more natural as society starts to cost environmental externalities (e.g. carbon, water and health benefits) that have traditionally not been included in accounting sheets.

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[6] Tucker (2006, p. 13): "*Of particular relevance here is the growing recognition that, despite numerous conservation initiatives and massive investments in conservation actions over the last 20 years, progress in conservation on the ground has been slow and erratic (….). As a result some major donors are pressuring NGOs for evidence on how they spend their money, how they learn and how well they have been achieving their aims*." See also the Conservation Measures Partnership (CMP www.conservationmeasures.org), a partnership of conservation organisations that are collaborating to develop and promote common standards and tools for designing, implementing, and measuring the impact of conservation actions.

What constitutes a policy relevant ecosystem value estimate? This is a difficult question to answer. One can though come up with a number of characteristics that meet the requirements of both science and practice for reliable value estimates. Firstly, according to all expectations, ecosystem value estimates should be *contextual* in the sense that they answer specific needs for a specific time and place. This in a sense runs counter to the need for generalization and transferability of value estimates across policy fields and/or countries, but on the other hand serves better the needs of a detailed and fine tuned environmental decision-making context such as the ones triggered in Europe by such ambitious legal documents as the *Water Framework Directive* and the *Environmental Liability Directive*.

Secondly, ecosystem value estimates should be as *accurate* as needed. In this respect, different policy contexts demand different degrees of accuracy spanning the range from detailed estimates for litigation purposes and full cost pricing to more general and instructive estimates of climate change impacts and "resource consciousness". An intermediate position could be reserved for estimates serving institutionalized payments for ecosystem services (Alpizar et al. 2007). The scaling of accuracy of value estimates according to the specific policy target served is, of course, a direct corollary of the non-trivial budgets necessary for the design and implementation of full-blown, state-of-the-art valuation applications.

Thirdly, the value estimates should correspond to the planning horizon of policy measures and the perceived time scale of impacts valued. There is a clear trade-off between precision and timing since both the inherently dynamic nature of ecosystems and our myopia for future impacts of present policies do not lend themselves to an easy and uncontroversial treatment of values over time (see section III.4 below).

Fourthly, the degree of aggregation is dependent on both the delimitation of the relevant population of beneficiaries (i.e. those whose preferences count) and the heterogeneity of the underlying socio-economic structure. An important issue in this respect is the empirically documented phenomenon of distance-decay according to which value expressions in stated preference surveys tend to get lower the further the respondent lives from the location of the object of valuation (Bateman et al. 2006). In any case, aggregating point value estimates to a wider total can be a tricky and dangerous enterprise as the follow up discussions of the Costanza et al. (1997) paper demonstrates (Toman 2003).

## II.3 What values, for what policy objectives?

Do people care for the environment? The published evidence for monetized expression of human preferences for conserving ecosystems, as it has been documented to date for individual biomes and the totality of the Earth's ecosystems, suggests the answer must be 'yes.' Pearce (2007) has questioned the validity of this assertion by comparing estimates of global ecosystem values with global expenditure on ecosystem protection. He notes that '*the world's willingness to pay for ecosystem conservation generally runs into many trillions of dollars, suggesting that the world does recognize the importance of ecosystem services and is willing to pay for them. But when we look at the actual expenditures on ecosystem conservation, they appear to be measured in, at best, a few billions of dollars annually. How can willingness to pay and actual payments differ by several orders of magnitude?*' (p. 314).

For people in the conservation arena this is a truly alarming assertion since it documents a ‘global deficit of care’ to resolve global warming and conservation problems.[7] For others it may simply indicate the shortcomings of value estimates and related economic methodologies. In the first case, valuing ecosystem services turns out to be irrelevant for policy purposes irrespective of being methodologically solid whilst in the second case it turns out to be methodologically flawed irrespective of being policy relevant. A possible explanation though might be linked to the dual nature of the valuation process: *estimation* and *capture*. The former answers the question of the magnitude of social benefits but the latter is actually concerned with translating the value estimates into behavioural changes. We turn our attention first to the problem of capture. How successful has the environmental decision-making process in the developed countries been to date in capturing ecosystem value estimates?[8]

Environmental management takes place amidst an array of non-economic and economic institutions. The most important of these are courts and markets. Courts outside the USA have rarely, if ever, given priority to economic valuation arguments especially when individual preference-based values are concerned. The new European Community Directive 2004/35 on environmental liability with respect to the prevention and restoration of environmental damage (known as the Liability Directive)[9] will bring before the European courts numerous compensation claims for natural resource damages. As a consequence, the implementation of the Liability Directive will require tools from the environmental sciences, especially environmental engineering and restoration ecology, which are essentially related to the diagnosis and restoration of ecosystem functions and services. This task will also require, for the first time, the systematic application of economic methods and tools (e.g. cost accounting and valuation of non-market goods and services) as well as the consideration of legal and administrative procedures in order to achieve the internalization of environmental externalities. In particular, the scientifically complex confrontation of the issues that arise from the implementation of the Liability Directive, in conjunction with the potentially significant economic externalities of industrial activities, will inevitably lead to judicial disputes and will require the expertise of a wide range of professionals. It is therefore expected to shed a new light on the conceptual, legal and moral appropriateness of non-market valuation methodologies.

To date, exclusion of preference-based economic valuation techniques in natural resource litigations has been advocated on several grounds (see Kontoleon et al. 2002): discrepancies between ex ante and ex post WTP, discrepancies between WTP and WTA welfare measures, redundancy of value estimates in determining restoration levels, inefficiency of value estimates in securing deterrence of potential liable parties, incongruity of legal and economic definitions of ‘legal standing’, and disproportionate demands on financial sources and expertise. Last, but not least, we note the assumed inappropriateness of using non-use values as they are likely to inflict unnecessary bias and risks on legal procedures.[10]

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[7] See though Turner and Daily (2008) for an optimistic account of the evidence on global conservation expenditures.

[8] This aspect of value estimation is discussed below in section III.

[9] The environmental liability Directive was due to be incorporated in national law systems by April 2007 but this is still not the case in a number of EU member states.

[10] See Kontoleon et al. (2002), Shavel (1993), Daun (1993), Boudreaux et al. (1999), Cummings and Harrison (1994), Hanemann and Keeler (1995).

Seen from a pragmatic perspective, the negligence of preference-based values in the courts has its counterpart in the controversies among economists related to the internal consistency of non-market value estimates. Many economists are very sceptical about stated preference methods and would rather advocate, inter alia, cost-based approaches, deliberative and inclusionary techniques, or multi-criteria tools for the measurement of ecosystem values. Moreover, it can be questioned if efficiency in the use of resources is a major pillar in the judicial approach to problem solving. 'Rights based' approaches, the power of 'status quo' as manifested in the benevolent treatment of historically invested property rights to resources and related concepts seem to influence judicial decisions disproportionally in relation to efficiency arguments.

We turn our attention now to the use of markets for capturing ecosystem values. While markets have the potential to trade ecosystem services at prices that reflect their value to society, they often fail to do so for reasons related to the characteristics of ecosystem services. Environmental policy is planned and implemented in the context of a constitutional mixture of conventional markets (e.g. timber, ecotourism, the production of anti-pollutant technology), hybrid markets (e.g. supply of drinking water), emerging markets (e.g. the market of transferable rights for emissions), and of completely absent markets (e.g. the protection of biodiversity, landscapes and nature) (Shogren 2002). Conventional environmental markets have been studied extensively under the general classification of 'ecosectors'. OECD, EUROSTAT, UNO, WTO, World Bank and professional associations document their trends, financial results and growth potential in conventional accounts.[11]

By definition, ecosectors internalize ecosystem services in their tariff structure and thus capture social values for services through their own policy objectives.[12] Hybrid (environmental) markets are characterized by the presence of a strong social component in their policy objectives and customer policy. These are the markets wherein public utilities operate, for example, drinking water supply in most European countries. These are also markets characterized by substantial deficits and therefore urgently in need of reforms. The European Community Directive 2000/60 on water resources management (known as the Water Framework Directive, WFD) is addressing the problem of value capture in this sector by institutionalizing the principle of full cost pricing of European water resources. The joint implementation strategy of the WFD expects that ecosystem services provided by water resources will be valued and suitably accommodated in EU water tariffs by 2010.

Emerging markets are increasingly entering the scene of conservation planning in a number of forms and objectives, e.g. GHG emissions, development rights, fishing allowances to mention a few.[13] A critical feature of any emerging market is how the commodity being offered for sale is defined. While commodities in existing markets are easily identified, this is often one of the most challenging aspects of ecosystem service market creation. It is also one of the most important steps for determining whether or not the market will take off and be sustained. Emerging markets for ecosystem services need to be viewed in the context of growing economies where the increasing sophistication of economic transactions have led to

[11] Carpentier et al. (2005) document the growing importance of ecosectors internationally and offer an illuminating analysis of the efforts and obstacles of integrating them in the overall strategy of the World Trade Organization.

[12] To be more precise, ecosectors internalize only a part of social ecosystem value. They fail to capture fully consumer surplus unless they are in the (unrealistic) position to practice a full-blown price differentiation policy and thus extract the maximum WTP from each costumer separately.

[13] See Gutman (2003), OECD (2002b), Binning et al. (2000), Pagiola et al. (2004).

instruments that offer wide opportunities for flexibility in how an exchange between two parties is achieved. Transactions in financial markets include complex instruments such as the buy/sell options or other financial derivatives (e.g. futures markets for oil and wheat). Some financial instruments are already in limited use in biodiversity markets. They have shown that ecosystem services can be more effectively and efficiently provided when transacted with the aid of financial instruments (OECD 2005).

In the context of emerging markets for ecosystem services, valuation may be needed to calibrate specific objectives. As a general rule though, prices for ecosystem goods and services are endogenously determined in the market place on a day-to-day basis. Over a period of time, the price of a financial instrument, such as a tradable permit, will reflect the marginal cost of technologies to substitute away from using the capped resource. For a biodiversity-related resource, the price of the permit will also reflect the marginal economic value of biodiversity loss. As such, the price can provide a continual basis for gauging whether the trade-off between biodiversity conservation and its loss is being made at a socially desirable level. Too low a price, for example, would suggest that biodiversity is being lost cheaply, and that the cap should be tightened. Valuation is thus an important tool whenever policies are being introduced to create or enhance marketability for goods and services not autonomously marketable (OECD 2005).

To recapitulate, valuing ecosystem services is relevant for policy purposes both in a non-market as well as in a market context. Notwithstanding unresolved problems related to the validity of value estimates, the use of non-market instruments by policy-makers should, in general, heighten the importance of valuation as a complementary tool. This is because policy that will make tradeoffs on everyone's behalf will need to gauge the wishes of all who have an interest in the issue. In many cases, some valuation will be necessary to underpin the policy.

## III. Issues in the estimation of ecosystem values

This section summarizes a number of basic concepts and tools within non-market valuation methods. It aims at highlighting those aspects of the methodologies that touch upon the main theme of this review, i.e. temporal dynamics and uncertainty issues. By doing so, it delineates the empirical part in section IV.

### III.1 Choices and values

Humans are specific in nature in that they possess a sense of 'right' and 'wrong', a by-product of their value system. 'Value systems refer to intrapsychic constellations of norms and precepts that guide human judgment and action. They refer to the normative and moral frameworks people use to assign importance and necessity to their beliefs and actions' (Farber et al. 2002). The above process of assigning importance and necessity is called valuation. Often valuation is used synonymously with evaluation: 'In common parlance, we value when comparing objects and evaluate when comparing the relative merits of actions' (Dasgupta 2001). The reason we have to value or evaluate is choice: 'The issue of valuation is inseparable from the choices and decisions we have to make about ecological systems' (Costanza 2000). In a world of finite (natural) resources, choosing among competing uses of these resources is imperative. Uses of resources (a wetland for example) can be direct (i.e. drinking water from the wetland) or indirect (i.e. protecting my house downstream from flooding), consumptive (cutting timber) or non-consumptive (enjoying the view of the wetland).

The criteria for choice can be manifold: economic, moral, cultural, aesthetic, ecological etc. The economic criterion of choice is tantamount to choosing the least cost option to achieve a certain utility level, or, in its dual form, choosing the maximum utility option to be achieved with a certain expenditure. An ecological criterion of choice (i.e. choosing which species to prioritize for protection) could be the degree of rareness. By the act of choosing we inevitably produce rankings, that is, relative values. Such values are always instrumental: 'We use the term 'value' to mean the contribution of an action or object to user-specified goals, objectives or conditions' (Costanza 2000). Alternatively, we define as intrinsic all those values that are disassociated from the concept of choice: items or beings possessing intrinsic value are to be preserved in their own right, irrespective of them serving any 'user-specified goals, objectives or conditions'. It is common in the environmental literature to identify instrumental values with anthropocentrism and intrinsic values with biocentrism. All values are quantified on the basis of a value metric (or numeraire): energy, money, or simply commodities.

Economic values for nature's services are characterized as subjective values because they are based on human preferences and quantified on the basis of the intensity of these preferences. The intensity of preferences is expressed in the amount of money an individual is willing to pay in order to enjoy a certain level of provision of nature's services (Willingness to Pay, WTP). Reversing the standpoint of the trade-off, the intensity of preferences can also be expressed in the amount of money an individual is willing to accept as compensation in order to tolerate a certain level of loss in the provision of nature's services (Willingness to Accept, WTA): 'The process of inferring preferences and estimating the willingness of individuals to sacrifice to achieve some outcome is termed 'valuation' (Armsworth and Roughgarden 2001). On the other hand, choices based on scientific criteria (i.e. the criterion of rareness mentioned above) produce what are conventionally called objective values (i.e. ecological values). Quoting from Webster's New World Dictionary 1988, Freeman (1997) asserts that 'I have found that economists and ecologists typically use the term 'value' (….) in two different senses when they use it in discussions of ecosystems. Ecologists usually use the term to mean 'that which is desirable or worthy of esteem for its own sake; thing or quality having intrinsic worth. Economists use the term in a sense more akin to 'a fair or proper equivalent in money, commodities, etc..., where 'equivalent in money' represents that sum of money which would have an equivalent effect on the welfare or utilities of individuals' (p. 241).

In instrumentally valuing a resource such as an ecosystem, the total economic value (TEV) can be usefully broken down into a number of categories.[14] The initial distinction is between use value and non-use value. Use value involves some interaction with the resource, either directly or indirectly: Indirect use value derives from services provided by the ecosystem. This might for example include the removal of nutrients, providing cleaner water to those downstream, or the prevention of downstream flooding. Direct use value, on the other hand, involves interaction with the ecosystem itself rather than via the services it provides. It may be consumptive use, such as the harvesting of reeds or fish, or it may be non-consumptive such as with some recreational and educational activities. There is also the possibility of deriving value from 'distant use' through media such as television or magazines, although whether or not this type of value is actually a use value, and to what extent it can be attributed to the ecosystem involved, is unclear.

[14] The definitions that follow are taken from FAO (2004).

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

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

- Bequest value is associated with the knowledge that a resource will be passed on to descendants to maintain the opportunity for them to enjoy it in the future.
- Philanthropic value is associated with the satisfaction from ensuring resources are available to contemporaries of the current generation.

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

- 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.
- Quasi-option value is associated with the potential benefits of awaiting improved information before giving up the option to preserve a resource for future use. It suggests a value in particular of avoiding irreversible damage that might prove to have been unwarranted in the light of further information. An example of an option value is in bio-prospecting, where biodiversity may be maintained on the off-chance that it might in the future be the source of important new medicinal drugs. It has been suggested that option value is less a distinct category of total value than the difference between an ex-ante perspective yielding 'option price' (consumer surplus plus option value) and an ex-post perspective giving expected consumer surplus, as a measure of value (Schmalensee 1972; see also Bishop 1986).
- Insurance value is conceptually linked to the above notions of option values: 'Identifying how close a system might be to collapse of some or all functions is itself extremely difficult, yet one would expect willingness to pay to avoid that collapse to be related in some way to the chances that the collapse will occur. If the chances are known, the value sought is then the premium that would be paid to conserve resilience' (OECD 2002a, p.31).

### III.2 Assets versus services

In the previous section we presented a general outline of the idea that (economic) values are products of choice. Deriving credible estimates of nature's social worth in contexts where there are either no apparent markets or very specific ones remains a practical problem associated with ecosystem valuation. One central merit of functioning markets is that they *define* the object of trade and its unit of measurement in a generally acceptable way. Contrary to this, ecosystem valuation has to rely on ad hoc definitions of what is being traded and its units. We face here again the 'endpoint problem' mentioned earlier as a problem of defining in concrete statements, intuitively expressed and commonly understood, what matters in nature. The recently proposed concept of the *Service Providing Unit* (SPU) seems to go a long way in supplying non-market valuation studies with a long sought after "ecological endpoint". An in-depth examination on this topic has been undertaken in another paper (Kontogianni et al. *in prep*.). Here, we confine ourselves to discussing the recent revival of the concept of ecosystem services by the Millennium Ecosystem Assessment and assess its importance for both valuation and policy.

For some commentators, the genesis of the ecosystem service concept can be traced back to Plato and his lamenting the degradation of forested hills in Attica (Daily 1997b, p. 5-6 based on Hillel 1991, Brauman et al. 2007, p. 69 based on Feen 1996). In a more detailed overview of the subject, Mooney and Ehrlich (1997) locate the beginnings of the modern use of the term in the year 1970 and assert that 'Two questions about ecosystem services have been clear from the start […]. One is how the loss of biodiversity will affect ecosystem services, and the other is whether it will be possible to find and deploy technological substitutes for the services' (p. 15). Today, several different definitions and classification systems are given in the scientific literature. Daily (1997b) p. 3 defined ecosystems services as: '*the conditions and processes through which natural ecosystems, and the species that make them up, sustain and fulfil human life.*'

Since then, a number of researchers have built on Daily's definition shifting the emphasis from functional (de Groot et al. 2002; Costanza et al. 1997; Hein et al. 2006) to organizational (Beaumont et al. 2007; Holmlund and Hammer 1999) to descriptive (Wallace 2007; Moberg and Folke, 1999; Boyd and Banzhaf 2007) characteristics (adapted from MEA 2003, ch. 2). In a recent attempt to clarify the seemingly contradictory definitions and classifications of ecosystem services, Fisher and Turner (2008) propose a flexible definition that takes into account the fact that both intermediate as well as final services are linked to human welfare and urge project scientists and stakeholders to *"...agree on the line between final services and benefits, so that we can manage, monitor and make policy to protect services that help maintain (and/or value) that benefit*" (p. 3). Here we follow the definition and classification given in the Millennium Ecosystem Assessment (2005), '*ecosystem services are the benefits people obtain from ecosystems'* which divides goods and services into four categories:

- Provisioning services are the products obtained from ecosystems, including food, fibre, fuel, genetic resources, ornamental resources, freshwater, biochemicals, natural medicines and pharmaceuticals.
- Regulating Services are the benefits obtained from the regulation of ecosystem processes including air quality regulation, climate regulation, water regulation, erosion regulation, water purification and waste treatment, disease regulation, pest regulation, pollination and natural hazard regulation.
- Cultural Services are the non-material benefits people obtain from ecosystems through spiritual enrichment, cognitive development, reflection, recreation, and aesthetic experiences, including cultural diversity, spiritual and religious values, knowledge systems, educational values, inspiration, aesthetic values, social relations, sense of place, cultural heritage values, recreation and ecotourism.
- Supporting services are those that are necessary for the production of all other ecosystem services. They differ from provisioning, regulating, and cultural services in that their impacts on people are often indirect or occur over a very long time, whereas changes in the other categories have relatively direct and short-term impacts on people. (Some services, like erosion regulation, can be categorized as both a supporting and a regulating service,

depending on the time scale and immediacy of their impact on people.) These services include soil formation, photosynthesis, primary production, nutrient and water cycling.

It is only recently that the analytical concept of ecosystem services and its corresponding concept of ecosystem assets was clarified and put in perspective within the environmental economics profession (Smith 1996b). In what is practically an introduction to his collected papers, V.K. Smith notes that '*conventional economic practice has dichotomized the analysis of natural and environmental resources. The former [...] have been treated as natural assets, thereby recognizing the important implications of each period's allocation decisions for the inter-temporal availability of resource's services. [...] By contrast, with environmental resources, the analysis has usually remained static*' (p. 7). Describing the problem thus in terms of asset utilization, V.K. Smith defines the value of natural capital as the net present value of its capitalized stream of services paving thus the road for a broad application of ecosystem services in the valuation literature.

The emphasis put on ecosystem functions and services showed in Europe in a number of consecutive and interlinked research projects on wetlands that have been carried out since the 1990's. Most notable are FAEWE II (CT95-0559) from 1994 until 1996, ECOWET (ENV4-CT96-0273) from 1996 until 1999 and EVALUWET (EVK1-CT-2000-00070) from 2001 until 2004 (Brouwer et al. 2003). These research projects already included important social science elements, more specifically the application of environmental and ecological economics principles in the domain of wetland ecosystem analysis and evaluation In FAEWE II, an important first step was made to link previous natural science work on a functional classification of wetlands, following the hydrogeomorphic (HGM) approach developed in the United States in the 1980s (e.g. Brinsson 1993), to the associated socio-economic values. This functional approach to sustainable wetland ecosystem management has dominated the three previously mentioned research projects in Europe (Brower et al. 2003).

### III.3 There is nothing simple about (environmental) preferences

The representation of individual preferences in economic theory as a (weak) ordering of alternatives has always been defended on account of rationality; a stance combined with a strong reluctance to dwell on the origin, dynamics, content and motives of human desires as integral characteristics of preferences. Accordingly, by keeping all substantial discussion on preferences outside the borders of economic science the ground was paved for a systematic and thorough examination of the formal properties of a specific prototype of preference structure, one characterized by a) uniqueness, b) stability and c) consistency. Consistency presupposes *inter alia* the property of transitiveness (Kreps 1990; MasCollel et al. 1995). Under these premises, the meaning of rationality and rational decisions has been studied and applied extensively in economics (Sudgen 1991; MacFaden 1999).

The recent tendency to substantiate preferences in a number of economic research fields (environmental economics, marketing, health economics) expresses the results of a fertile dialogue between psychologists and economists concerning the nature of individual preferences and their role in micro decision-making processes (Smith 1991; Kahneman and Tversky 2000). Results from what has come to be termed 'behavioural economics' prompt us to adopt alternative hypotheses in order to accommodate the existence of empirical 'anomalies' within the corpus of formal neoclassical consumer theory (Starmer 1999; 2000; Shafir and LeBouef 2002; Camerer 1995; Goldstein and Hogarth 1997).

The points of critique are, as expected, of unequal weight and importance: the theoretical edifice of neoclassical theory of choice assimilates the problem of non-unique, multiple preferences by recognizing that different institutional contexts (market versus political processes) induce (or constrain) individuals to act on the basis of different value and preference systems.[15] The phenomenon of 'multiple selves' emerges which, in its most interesting form, splits each one of us into a consumer and a citizen.[16] The motives of individual preferences are accordingly very complicated; faced with choice dilemmas in situations where public, environmental goods are at stake, '*...people were quite ready to take moral responsibility and act as citizens as well as consumers*' (Kontogianni et al. 2003, p. 328).

The problem of preference stability seems also not decisive if by that we mean stability through time: it is obvious that individual personalities evolve and their preferences are accordingly modified. A potentially more severe problem though has to do with statistical stability of choices in repeated choice experiments in which choice rankings seem to change in consequent rounds violating often the hypothesis of transitivity. We summarize hereafter the main points of this last empirical 'anomaly' of choice. Put in general terms, individuals often ask for a higher price in order to sell a good than they are willing to pay in order to obtain it. Within the context of hypothetical markets, the problem takes on the form of WTP / WTA disparity.[17] This is known in the literature as the 'endowment effect' or 'status quo bias' (Thaler 1980; Samuelson and Zeckhauser 1988).

Since the 1980s, a host of empirical studies - mainly choice experiments and contingent valuation applications – have attempted to understand the true nature of this 'anomaly' and test its persistence under conditions of repeated (hypothetical) market transactions. In the case of the endowment effect where the problem is associated with our inexperience *vis a vis* transactions which involve environmental goods, it can be expected that repetition and learning would finally resolve it. Rationality would again be restored in such markets and neoclassical choice theory would have been capable of integrating it within its corpus. In the case where the problem is due to the well known tendency of interviewed individuals to use strategic responses of the type 'buy cheap, sell expensive' then it should again tend to resolve itself in contexts of hypothetical markets where the possibility of punishment (loss) of irrational responses were present. Alas, in both cases the 'anomaly' seems to persist (Knez et al. 1985; Coursey et al. 1987; Knetsch and Sinden 1984; Frey and Eichenberger 1994).

Let us translate the 'endowment effect' in terms of individual gains and losses. It becomes obvious then that individuals consider reductions in their own initial endowments (their material status quo) less profitable than the advantages derived from an equal growth in the same initial endowments. We are forced to admit that this entails an *asymmetry of subjective values*, which in turn seems to be grounded in a natural aversion towards losses from an established point of reference (the status quo). The phenomenon is known today as 'loss aversion' (Kahneman and Tversky 1984).

The value asymmetry described above leads to the revision of a number of established principles in neoclassical choice analysis. To start with, the way that individuals perceive

[15] For a state-of-the-art discussion, see Russell et al. (2003).

[16] The terms 'consumer' and 'citizen' in this context are based on Sagoff (1988). The essential difference between the two is related to the degree of preponderance of public versus private interests in individual decision making.

[17] Empirically estimated in the analogy 1 to 2.

changes in their own utilities in different states of the world A and B cannot be simply the comparison of the respective end states A and B but the comparison of the quantities 0±A and 0±B, that is relative changes from a neutral, reference point 0, where 0 stands for the status quo. This is the case not only in situations of risky choices but even in riskless choice situations (Tversky and Kahneman 1991). Another way to express the value asymmetry is to admit that opportunity cost is a totally different concept from real, out of pocket cost (Kahneman et al. 1990). This last idea points to the importance of fairness perceptions referring to the initial distribution of property rights (Kahneman et al. 1986). If the above analysis is correct, choices cannot be (at least conceptually) reversible. The 'distance' in utility terms between an initial position 0+A and the subsequent position 0 (that is, the losses incurred through giving A away) does not equal the 'distance' between an initial position 0 and 0+A (that is, the benefit incurred through obtaining A). Value asymmetry leads therefore to indifference curves that intersect! (Knetsch 1990).

Faced with the above described 'anomalies' of mainstream economic analysis of choice, and the consequent treatment of preferences in a formalistic mode, devoid of any substantial content, a number of economists turn their attention to alternatives that analyse human preferences in a simulated social context (Wilson and Howarth 2002; Jacobs 1997; O'Neill and Walsh 2000; Sagoff 1998; Spash 2007). An interesting version of deliberative techniques is the 'group Contingent Valuation'. With a group CV, the explicit goal is to derive an economic value for the ecological good or service in question. The valuation exercise is conducted in a manner very similar to a conventional CV survey - using hypothetical scenarios and payment vehicles - with the key difference being that value elicitation is not done through private questioning but through group discussion and consensus building. Such deliberative or inclusionary techniques are more important in situations where the issue at hand is loaded with distributional conflicts and moral aspects.

### III.4 A digression on dynamics, uncertainty and informational gaps

We stress at this point, as emphatically as possible, that choice in the realm of ecosystem conservation is always a complex task. Ecosystems are complex systems of biotic and abiotic components. As such, ecosystems are also notoriously dynamic systems. Their complexity is mirrored in the multiplicity of their forms being responsible for both their high primary productivity and their vulnerability to exogenous shocks. Their dynamic nature leads both to periodic and cyclic as well as to progressive change. Under natural conditions, the state of the coast, for example, represents '*an optimal but ephemeral dynamic response to an equilibrium that exists between the material form of the coast and the 'forcing factors' of waves and tidal currents*' (Crooks 2003, p. 8).[18]

What does the recognition of ecosystem complexity entail for the economic approach to valuation? This is a crucial question since a lot of criticism on the latter is emanating, unavoidably it seems, from the former. From the beginning of economic thought, economists have taken for granted the analytical legitimacy of drawing very simplified pictures of the natural world when investigating industrial production and the consequences of economic behaviour. Today's environmental crisis reveals the fact that such purposeful abstractions from the concrete material basis of production, necessary as they may have been occasionally within social analysis, can prove very dangerous when used for policy prescriptions. It is this contradiction between the notion of abstract, quantifiable economic value and the concrete,

[18] For a recent state-of-the-art review on the concept of dynamic ecosystems see Vanderwalle et al. (2008).

limited and qualitative *Physis* that went, with rare exemptions, unnoticed in the development process of economic tools and techniques. Classical political economy, with its main representatives Adam Smith, David Ricardo and Thomas Robert Malthus has always been striving to keep the memory of pre-industrial, concrete wealth of *oeconomia natura* alive but could not at the end resist the sweeping force of market reality and neoclassical formalism (Skourtos 1998).

Meanwhile, fundamental changes have occurred in our understanding of ecosystem functions and services, and these have prompted many recent international efforts to protect and sustainably use them. From their own perspective, and in the face of this widespread recognised oblivion of economic theory, modern environmental economists have been engaged in an in-depth re-examination of the hidden assumptions and tacit allowances about the natural world that support the main conclusions of their policy prescriptions. Thanks to joint efforts with natural scientists, our 'production functions' linking natural and engineering processes with economic goods and services are far better understood to date as are also, though to a lesser extent, their unintended consequences, negative externalities or 'bads'. In spite of scientific advancement though, the remaining gaps in our knowledge loom large. In an authoritative review of valuation approaches to aquatic ecosystem services, it is emphatically noted that '*comprehensive valuation of aquatic ecosystems should be viewed as a practical impossibility*' (NAS 2005, p. 87). The authors nevertheless stress the fact that '*comprehensive ecosystem valuation is not generally essential to inform many management decisions*' (p.89). A way out of the ensuing dilemmas lies with consolidating and refining available integrated approaches to environmental planning while backing decisions with proper stakeholder involvement. (Harremoes and Turner 2001)[19]

How could we describe the task of ecosystem valuation under dynamic change and uncertainty? How could we link ecosystem changes to human welfare in an evolving, non steady state and coupled economy-ecology system? Are valuation methodologies inherently static? Dynamic values and preferences are specific aspects of a more general inquiry in economics known as the intertemporal allocation problem (Heal 1986). Research on the intertemporal allocation problem in natural resource economics focuses mainly on the optimal time path of using renewable resource stocks. For the purposes of the present review, we will discuss a modern variant of the intertemporal allocation problem in the form of dynamic simulation models (sections IV.3.1 and IV.3.2). The rest of our analysis is based on a less 'authentic', though more practical way of analysing dynamic phenomena, the so-called comparative static approach to the dynamics of value estimates (sections IV.2.1, IV.2.2)

In order to illustrate the problem of value dynamics in a coherent framework we use the following simple model (based on McConnell 1990 and Jakus et al. 2006). Let us define:

The *indirect utility function* V as the function denoting the maximum utility level $u_{max}$ achievable at given prices $\mathbf{p}$, income y and ecosystem services $q^0$:

$$V = u_{max} = v(\mathbf{p}, q^0, y) \qquad (1)$$

[19] In more technical terms, Levin and Pacala (2003) argue that '...the most reliable approach to guiding decision making is through the development of robust simulation models, which can be the basis for informed scenario development' (p. 84).

The *expenditure function* E as the function denoting the minimal amount of income $y_{min}$ needed to achieve a given utility level $u^0$ at prices **p** and ecosystem services $q^0$:

$$E = y_{min} = e(\mathbf{p},q^0,u^0) \qquad (2)$$

The *compensating variation* C as the change in a person's income that will make him indifferent between two situations with unequal provision of ecosystem services, $q^0$ and $q^1$ and expenditures (incomes) $E^0$ and $E^1$. This is what answers in a stated preference survey are supposed to reveal: the maximum willingness to pay (minimum willingness to accept compensation) in order to secure a gain in ecosystem service provision (accept a loss in ecosystem service provision). C can accordingly be expressed as the difference between two expenditure functions achieving the same utility level with different levels of service provision:

$$C = e(\mathbf{p}, q^0, u^0) - e(\mathbf{p}, q^1, u^0) \qquad (3)$$

Substituting the indirect utility function (1) into (3), we have:

$$C = e[\mathbf{p}, q^o, v(p, q^0, y)] - e[\mathbf{p}, q^1, v(p, q^1, y)] \qquad \text{or}$$
$$C = g(p, q^o, y) – g(p, q^1, y) \qquad (4)$$

We denote the right hand side of equation (4) as the *variation function* s:

$$S = s(\mathbf{p}, q^0, q^1, y) \qquad (5)$$

The variation function denotes the compensation needed to hold utility constant with changing ecosystem service provision from $q^0$ to $q^1$. In its general form, the variation function will include as arguments - besides prices, income and the quantity of the public good – any variable believed to influence C. Such variables can, for convenience, be subsumed in the general term "socio-demographics" D. If we further index our variables with a time dimension t, then we have the dynamic version of the variation function as:

$$S_t = s(\mathbf{p}_t, q_t^0, q_t^1, y_t, D_t) \qquad (6)$$

Assuming a linear functional form for s(.), an econometric estimation of (6) will yield regression coefficients β that reflect the original preference structure. We can then define changes of value estimates through time (expressed as WTPs) for periods $t_0$ and $t_1$ as follows:

$$\Delta WTP = s(\mathbf{p}_0, q_0^0, q_0^1, y_0, D_0 : \beta_0) - s(\mathbf{p}_1, q_1^0, q_1^1, y_1, D_1 : \beta_1) \qquad (7)$$

We are now in a position to better classify and analyze dynamic aspects of ecosystem value estimates. In order to do this, we define as *demand-driven value dynamics* the phenomenon of value changes due to factors affecting the demand side of ecosystem services. These factors could be *inter alia* changes in income y, prices of other goods **p**, and the socio-economic profile D. We discern two aspects of demand-driven value estimates: the short to medium term variability of values, a point we address in section IV.2.1, and the long term variability of values, a point we address in section IV.2.2. We also define the *supply-driven value dynamics* as the phenomenon of value changes due to changes in the supply of ecosystem services q, even if (although not probable) preferences β, income y and socio-economic profiles D remain constant. We address a number of different approaches to this topic in section IV.3.

### III.5 A typology of methods

A number of economic valuation methods have been developed in order to cope with the complexity and variety of policy contexts, which delineate each ecosystem valuation application. It is customary to classify the methods into three categories (Freeman 2003, Braden and Kolstad 1991):

- Real Market approaches: costing in a real market context of projects and technologies that restore ecosystem service degradation (i.e. restoration cost approach).
- Surrogate market approaches: Revealing preferences for ecosystem services through actual choices in markets linked to the service in question (i.e. travel cost approach).
- Simulated market approaches: Stating preferences for ecosystem services in a hypothetical market setting (i.e. contingent valuation method).

Table 1 gives an overview of the main valuation methods in use today.

For the purposes of the present review, methods have been categorized into two classes:

- Methods that treat ecosystem services as (final) objects of consumption.
- Methods that treat ecosystem services as productive inputs to (final) objects of consumption.

This classification highlights the distinction between *demand-driven* and *supply-driven value dynamics,* and enables a separate focus on parameters affecting the demand (i.e. income y, prices of other goods **p**, socio-economic profile D, preferences β) and factors affecting the supply of ecosystem services (i.e. q).

## IV. Reviewing the empirical evidence

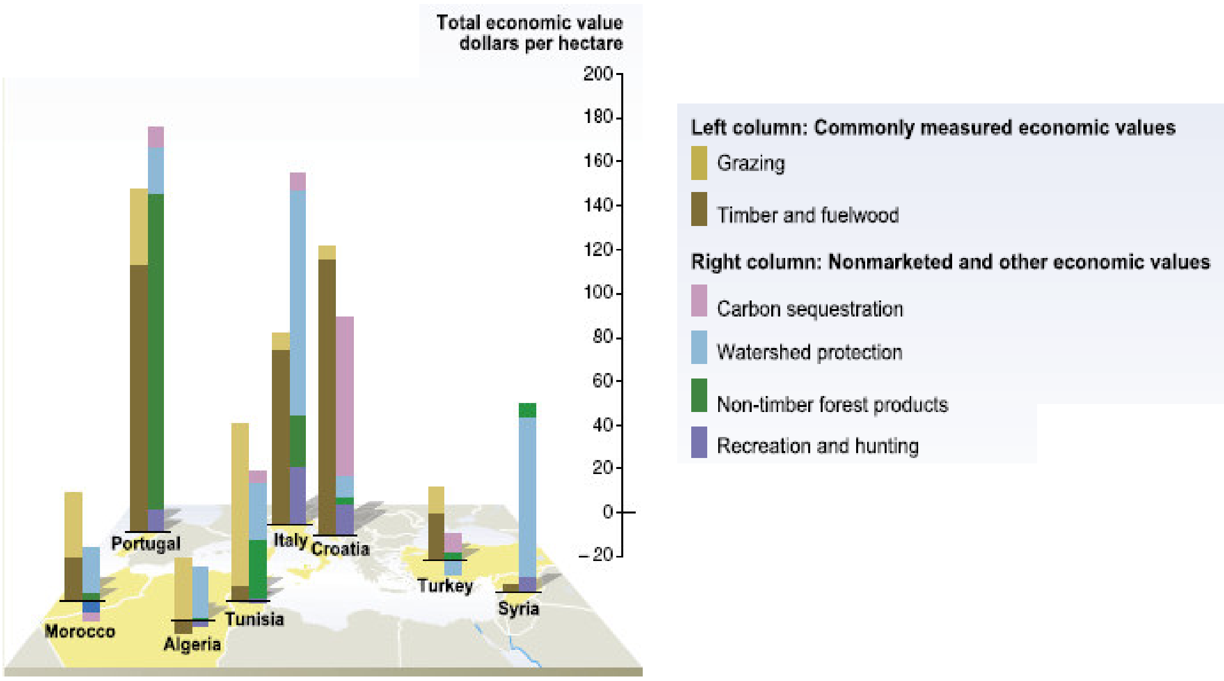
There has been much work and many reviews on the use of economic, preference-based approaches to estimating the social value of ecosystem services. Some recent reviews include: EFTEC (2005), which reviewed the economic, social and ecological value of ecosystem services focussing on wetlands, forests and agroecosystems; Kettunen and ten Brink (2006), which reviewed the cost of biodiversity loss in Europe; Moran et al. (2007), which determined a monetary estimate of the environmental benefits derived from the implementation of the nature conservation measures in the proposed UK Marine Bill; Ledoux and Turner (2002), which assessed the non-market values of coastal ecosystem services; Kaval (2007), which reviewed estimates for the recreational benefits of US Parks; and Madureira et al. (2007), which reviewed value estimates for agricultural, non-commodity outputs in Europe. The Canadian valuation database *ENVRI,* the most populated valuation database to date, hosts 2088 valuation studies of which 1168 refer to Northern America. As an example, Figure 2 shows the spatial allocation and magnitude of forest ecosystem values, as shown in MEA (2005).

**Table 1: Economic valuation methods.**

| Method | Description | Use values | | Non use values |
|---|---|---|---|---|
| | | Direct | Indirect | |
| Market analysis | Where market prices of outputs (and inputs) are available. Marginal productivity net of human effort/cost. Could approximate with the market price of a close substitute. Requires shadow pricing. | √ | √ | |
| Productivity losses | Change in net return from marketed goods: a form of dose-response market analysis. | √ | √ | |
| Production functions | Functions treated as an 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 pricing | 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 | Cost incurred in reaching a recreation site as a proxy for the value of recreation. Expenses differ between sites with different environmental attributes. | √ | √ | |
| Contingent valuation | Construction of a hypothetical market by direct surveying of a sample of individuals and aggregation to encompass the relevant population. Problems with potential biases. | √ | √ | √ |
| Damage costs avoided | The costs that would be incurred if the ecosystem function were not present. | | √ | |
| Defensive expenditures | Costs incurred in mitigating the effects of reduced environmental quality. Represents a minimum value for the environmental function. | | √ | |
| Relocation cost | Expenditure involved in relocation of affected agents or facilities: a particular form of defensive expenditure. | | √ | |
| Replacement/ substitute cost | Potential expenditures incurred in replacing the function that is lost; for instance by the use of substitute facilities or shadow projects. | √ | √ | √ |
| Restoration cost | Costs of returning the degraded ecosystem to its original state. | √ | √ | √ |

Source: Turner et al. (2001b).

**Figure 2: Annual flow of benefits from forests in selected countries. Source MEA (2005).**




In comparison there has been very little work relating to the dynamics of values and preferences or other aspects of complexity and uncertainty. To place the issue of value dynamics in the MEA terminology, the temporal dimension of social benefits derived from ecosystem services vary from direct, short to medium term benefits (provisioning) to indirect, medium to long term benefits (regulating), to direct, long term benefits (cultural), to indirect, long to very long term benefits (supporting). The last category of long to very long temporal benefits is what some researchers would prefer to call ecological benefits in contrast to the short to medium term socio-economic benefits.

To our knowledge, the present review is the first attempt at systematizing what we know about dynamic ecosystem values and preferences. This is not an easy task, for empirical valuation studies for ecosystem services often appear scattered throughout the scientific literature and are uneven in quality. In addition, issues of dynamics in value estimation are intertwined with all sorts of related issues on discounting future values and the presence of thresholds. Despite these difficulties, we attempt here a non-exhaustive review of the existing literature on dynamic aspects of ecosystem valuation in order to provide useful insights for further research in the area. Such an exercise provides to the interested reader a sense of the established approaches and possible future paths for ecosystem valuation.

## IV.1 Materials and methods

Our interest in collecting and reviewing material on ecosystem valuation lies more with the methodological than the quantitative aspect. As the references in the previous section indicate, quantitative assessments of value estimates abound in the literature. It makes no sense, in view of the thin and multifaceted inventory of dynamic value estimations to attempt a primarily quantitative assessment. Accordingly, we review here primarily the various approaches to capture the dynamic aspect of value estimates.

All information presented below was obtained from studies that were published mostly after 2000 in peer review journals. The literature search involved an intensive review of databases on the World Wide Web available at the University of Aegean. A specific data-base in MS ACCESS© was created. Several keywords were applied for the search: ecosystem services, ecosystem values, temporal reliability of values, value dynamics, evolution of preference, intertemporal valuation. Our selection criterion has been the study's explicit treatment of dynamic aspects of value estimates.

This search yielded 183 citations. Each citation was then located and reviewed by the authors. In cases where a paper addressed multiple values and services, a single citation provided more than one entry in our database. 153 citations (84%) were not examined further because they did not explicitly address the dynamic aspects of ecosystem valuation. The literature review yielded a total of 30 studies for further analysis and discussion. Results from these studies were then sorted solely by methodology. On this basis, each study was classified as addressing either demand-driven or supply-driven dynamics. Selected valuation estimates and methodologies are reported in detail below in sections IV.2 and IV.3.

### IV.2 Demand-driven value dynamics

Demand-driven value dynamics refer to short to medium term changes in the ecosystem values due to the influence of demand parameters $\mathbf{p}$, $y$, $D$ and $\beta$ in the variation function (7). We turn our attention first to 'test-retest' reliability applications before we focus on their long term evolution.

#### IV.2.1 Temporal reliability

Temporal reliability of value estimates is a phenomenon that potentially has considerable policy relevance for short to medium term decisions. Within this time period, human preferences could be considered as stable; the variance in value estimates could then be attributed to changes in the factors affecting WTP as they are modelled in the bid functions. Interest in short to medium term stability of value estimates has its origins in the assertion of the NOAA Panel that: "*Time dependent measurement noise should be reduced by averaging across independently drawn samples taken at different points in time. A clear and substantial time trend in the responses would cast doubt on the "reliability" of the findings*" (Federal Register, January 15, 1993, 4609). As Carson et al. (1997) noted, the reasoning underlying the Panel's call for the "temporal averaging" of WTP responses obtained from CV surveys as one method for increasing their reliability is not clear. A possible explanation could be the fact that WTP estimates are rather inflated if measured immediately after an event, such as an oil spill. Logically, '*the Panel's suggestion might be treated as a concern over the timing of a single CV survey in relation to the event giving rise to natural resource injuries*' (Carson et al. 1997, p. 152).

Table 2 presents a summary of 18 CV studies identified, which address the problem of temporal reliability of value and/or bid function estimates. Not all studies preserve the central assumption of test-retest methodology: same survey instrument, same sample, in two different points in time. For example, Whitehead and Hoban (1999) employ the same survey instrument with different samples and find that WTP estimates are significantly different over a five-year time period. But after controlling for factors that affect WTP the differences are not statistically significant. Zandersen (n.d.) employs the same format and sample to conclude that determinants of WTP for forest attributes have clearly changed between 1979 and 1999.

**Table 2: Temporal reliability studies. Source: Adapted from McConnell et al. (1998).**

| Authors | Test | Time elapsed between applications | Good/Service | Reliability |
|---|---|---|---|---|
| Loehman and De (1982) | Correlation between responses | 3 weeks | Health effects of pollution | Confirmed: high correlation |
| Jones-Lee et al. (1985) | Equality of responses | 1 month | Reduction in risk of accidents | Confirmed: no significant differences |
| Kealy et al. (1988) | Paired differences of WTP | 2 weeks | Chocolate bars | Confirmed: no significant differences |
| Loomis (1989) | Paired t-tests for equality of WTP | 9 months | Water levels in Mono Lake, California | Confirmed: coefficients and WTP stability |
| Kealy et al. (1990) | Equality of preferences | 2 weeks | Private good: chocolate bars. Public good: de-acidification of lakes in New York | Confirmed: coefficients stability and high correlation between responses |
| Reiling et al. (1989) Reiling et al. (1990) | Equality of WTP between distinct samples | On season versus off | Black fly control in Maine | Confirmed: WTP stability |
| Musser et al. (1992) | Correlation between responses. Equality of WTP. No carryover effect | 13 months | Cross country skiing in the National Forests of Oregon | Confirmed: WTP stability, high correlation, and no significant differences with respect to control groups |
| Teisl et al. (1994) | Correlation between responses | 12 months | Moose hunting permits | Confirmed: high correlation |
| Epp and Gripp (1994) | Correlation between WTP. No carryover effect | 6 months | Tropical forest preservation | Confirmed: high correlation, no significant differences with respect to control groups |
| Carson et al. (1997) | Equality of parameters between distinct samples | 4 years | Oil spill protection | Confirmed: Stability of equation parameters |
| Whitehead Aiken (2007) | Equality of WTP between distinct samples | 5 years | Bass and trout fishing, deer hunting and wildlife watching | Not conclusive: WTP declined but a clear reason was not detected |
| Berrens et al. (2000) | Equality of WTP functions | 1 year | Instream flows | Confirmed: temporal stability of WTP and bid functions |
| Brouwer (2006) | Paired t-tests for equality of WTP | 6 months | Health effects | Confirmed: temporal stability of WTP and bid functions |
| Zandersen (n.d.) | Paired t-tests for equality of WTP | 20 years | Forest recreation | Not confirmed: Both coefficients and mean WTP differ substantially |
| Downing and Ozuna (1996) | Equality of WTP functions | 1 and 2 years | Recreational fishing | Not confirmed: Both coefficients and mean WTP differ substantially |
| McConnell et al. (1998) | Equality of WTP. No carryover effect | 2 months | Recreational fishing | Confirmed: temporal stability of WTP |
| Whitehead and Hoban (1999) | Paired t-tests for equality of WTP | 5 years | Air and water quality | Confirmed: temporal stability of WTP |
| Jakus et al. (2006) | Paired t-tests for equality of WTP | 4 years | Recreational hunting | Confirmed: temporal stability of WTP |

Brouwer (2006) employs the same survey instrument in two different samples to conclude in favour of WTP stability within a 6-month period characterized by the occurrence of an extreme event in the second period. McConnell et al. (1998) employ the same survey instrument with different samples to test for the stability of WTP estimates and bid functions for recreational fishing in a two-month period. Nevertheless, in the face of considerable differences in the time span examined (2 weeks to 20 years) it is difficult to be conclusive about WTP values being stable over time. The evidence shows that this is the case for time periods less than a year and certainly not the case for time periods of twenty years. For periods of four to five years the evidence is mixed.

### IV.2.2 Evolution of preferences

The necessity of acting now on behalf of ecosystem protection is often based on the explicit assertion that "the very distant future" holds huge but discounted values of ecosystem services. Under the sustainability principle there is a requirement for the sustainable management of environmental resources, whether in their pristine state or through wise use, to ensure that the legacy of our current activities does not impose an excessive burden on future generations. However, the shift in emphasis from natural assets to ecosystem services (see section III.2) injects conservation management strategies with a certain degree of freedom: the objective is now the conservation of specific services not specific natural assets. As the admittedly extreme debate on "plastic trees" in California in the 1970s has demonstrated,[20] societies may opt to sacrifice specific natural assets in the long run if the ecosystem services supplied by them can in part or *in totto* be supplied by a new source. This contradicts fully the suggestion that it is 'large-scale complex functioning ecologies' that ought to form part of the intergenerational transfer of resources (Cumberland 1991).

From the standpoint of the present generation, conservation of natural resources impinges on posterity through two distinct paths: on the one hand, we influence future availability of natural resources by assessing future values for future stocks. On the other hand, we do the same by assessing present values for future stocks. The latter approach is what we call *discounting*. The temporal scale, combined with the discount rate, influences the present value of the streams of costs and benefits. Deep-seated disagreement prevails on the proper exchange rate among competing generations. The present generation may be future averse or future prone, therefore exhibiting a range of time preference rates. In practice though, the scientific community seems to have reached an agreement on proper rates of discounting future costs and benefits: the proper discount rate to use being one that reflects the opportunity cost of capital.[21]

The former problem is a rather neglected but genuine dynamic problem of values and preferences: the assessment of *future* preference values for *future* stocks (Horowitz 2002). Not *our* preferences for *their* stocks (the discounting problem) but *their* preferences for *their* stocks are the subject of our inquiry under the heading of *preference evolution*. This problem is distinct from the one discussed in section IV.2.1 not only because of the longer time periods

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[20] We refer to the debate of whether 'planting' plastic trees is a suitable substitute for natural trees removed for the need of public works in San Francisco. The still timely debate is contained in Stone (1974).

[21] Portney and Weyant (1999), summarizing the results of a workshop state: "Virtually everyone agreed on a standard procedure for the evaluation of projects with timeframes of forty years or less. Specifically, they agreed not only that it is appropriate to discount benefits and costs for the purposes of making present value comparisons, but also that the discount rate to use should be one that reflects the opportunity cost of capital" (p. 7) as cited by Horowitz (2002).

involved but also because now neither the form of utility function or its coefficients can be considered as stable.

Stable preferences could be assumed in the limiting case where we are only interested in the dynamic relationship between rising incomes and natural scarcities with future values of ecosystem services. It is a well-established result from microeconomic demand analysis that income is positively related to WTP amounts causing an upward shift in the demand schedule of a representative consumer. And of course, rising scarcity of ecosystem services causes an upward movement in the demand schedule. If, according to all evidence, incomes in the future are rising and natural assets are becoming scarcer, then is logical to assume rising future values of ecosystem services. We could call this a *weak* form of preference evolution.

A strong form of preference evolution involves a change in the preference structure of individuals. We discern again two different aspects here. The first one could be described as a (short to medium term) *learning* procedure through which individual preferences are shaped (or, if you prefer, constructed). A growing number of stated preference valuation studies demonstrate the mechanics of this learning procedure through adequate design of elicitation formats. In a recent paper, Bateman et al. (2008) apply what they call the 'learning design contingent valuation' (LDCV) method in order to investigate the formation and nature of preferences for unfamiliar goods such as ecosystem services. The authors address mainly two questions: (i) what is the speed at which individuals can form stable preferences for relatively novel goods presented in unfamiliar markets? and (ii) are those stable preferences, once formed, consistent or at variance with standard theory? The findings suggest that in such cases preferences converge towards standard expectations through a process of repetition and learning. Looking at the same results from the point of view of the dynamic aspects of preference and value formation, we may interpret the process of learning investigated by Bateman et al. (2008) as a strong form of preference evolution.[22]

The second aspect of strong preference evolution could be described as a (long term) change of preferences either through cultural transmission or evolutionary adaptation. Models of cultural preference transmission assume that preferences are acquired through an adaptation and imitation process. This process depends on parents' socialization actions and on the cultural and social environment in which children live (Bisin and Verdier 2001, Bowles 1998). Models of evolutionary adaptation instead follow an alternative, more recent approach by exploring the evolutionary foundations of preference formation. This approach, by focusing on evolutionary selection mechanisms, shows that in a variety of contexts individuals can actually obtain higher payoffs if they strive to maximize some distorted form of their actual payoffs (Heifetz et al. 2007) This leads us directly to the discussed 'anomalies' in the neoclassical theory of individual choice (section III.3), which now are seen as integral characteristics of individual rationality bounded by uncertainty, available heuristic rules and altruistic motives (Aversi et al. 1999).

Moreover, this line of research suggests that institutions themselves may influence the long term preferences of participating agents. Norton et al. (1998) emphasize the endogeneity of preferences and assert that in the face of changing preferences the old assumption of 'consumer sovereignty' is not adequate. They offer a coevolutionary explanation of preference evolution which, according to the authors, reconciles both the cultural transmission and the evolutionary approaches discussed earlier: '*From these sources we conclude that*

[22] See also List (2003), Shogren (2002) and Plott (1996).

*preferences are formed in humans (and many other animal species) by selection acting on traits that are transmitted both genetically and (in the case of humans) culturally, in a coevolutionary way'* (Norton et al., p. 201). Exploring the extent to which preferences may be in part an endogenous feature of the particular institutional framework seems to present promising avenues for future research.

## IV.3 Supply-driven value dynamics

Supply driven dynamics are an inherent feature of long term aspects of ecosystem service valuation influencing the supply parameter q in the variation function (7). We focus here on two closely related classes of models applied to supply driven value dynamics, the integrated simulation and dynamic bioeconomic approaches.

### IV.3.1 Integrated simulation models

A number of recent valuation studies are based on the notion of socio-ecological systems and address explicitly the complex interactions and feedbacks between human and natural systems (Liu et al. 2007, Scoones et al. 2007). Such integrated models apply simulation techniques to characterize the time path (co-evolution) of key variables, among them value estimates or *prices*, under various ad hoc assumptions concerning the state variables. The interest in the dynamic aspects of ecosystem valuation is fuelled by the recent proliferation of analytical attempts to formalize the complex interrelationships of joint socio-ecological systems based on the notions of resilience, stability, durability and robustness (Limburg et al. 2002, Scoones et al. 2007, D'Alesandro 2007, Li and Löfgren 1998, Walker et al. 2004, McPeak et al. 2006).

One of the first applications of a joint ecological economic simulation model was undertaken in order to evaluate the cost of plant invasions in the fynbos ecosystems of South Africa (Higgins et al. 1997). The authors developed a dynamic ecological economic model which values the ecosystem services provided by fynbos ecosystems under different management regimes. The ecosystem services valued were: water production, wildflower harvest, hiker visitation, ecotourism, endemic species and genetic storage. A hypothetical fynbos ecosystem was modelled with the help of five interactive sub-models, namely hydrological, fire, plant, management and economic valuation, and their interactions. For reasons of tractability, a spatially aggregated structure and a monthly time step was selected in order to simulate the seasonal dynamics of the system. Unit values of the ecosystem services examined were approximated on the basis of expert judgements and replacement cost. The unit values were assumed constant in time. For a time period of 50 years and a discount rate of 3% the scenario analysis showed that the ecosystem net present value varied from US$4.2 million (under a low valuation and poor management scenario) to US$66.7 million (under a high valuation and good management scenario). The value dynamics of fynbos ecosystem services in this application is therefore a genuine supply-driven dynamic, wholly dependent on the supplied quantities.

Chopra and Adhikari (2004) modelled the joint socio-economic system of a wetland in Northern India, designated as a Ramsar site and a national park, in a similar way. A dynamic simulation model in a 'STELLA' environment was set up to understand the linkages between underlying ecological relationships and economic values emerging from them. The model consists of four modules: the water module, the biomass module, the birds module and the net income module. The last module sums up the impact of changes in each of the preceding modules on income from tourism and resource extraction. The model applies a modified travel cost approach to derive the sensitivity of tourist visits to ecological health indices in a series of simulated scenarios with respect to future pressures on the park. It also applies net income calculations to assess the indirect local benefits from tourism. It runs on a monthly time step for a period of 27 years (1983 to 2010). The simulations point towards a critical dependence of economic value (direct and indirect income derived from the park) on ecological health indices. As the authors note, *"...a less attractive bird habitat reduces both number of tourists and the responsiveness of their numbers to further changes in the habitat attractiveness. Perhaps the arrivals get limited to the hard core of enthusiasts*" (p. 35). This example is also a supply-driven dynamic of values, wholly dependent on the quality of the supplied services.

The most representative piece of work in this area is probably Winkler (2006a and b). In two complimentary papers, R. Winkler sets out to formalize the chimera of complex, integrated socio-ecological system dynamics and *"...to provide the conceptual foundations for a new method of valuation of ecosystem services, which deals simultaneously with the ecosystem, the economic system and society in a balanced way*." With the help of a simplified, pre-industrial model (set up in Winkler 2006a and applied further in Winkler 2006b), the author aims to show how the interdependencies between the three subsystems influence values and how values change over time.

A distinctive element in Winkler's model is the formulation of a societal 'value system'. The specification of the 'value system' necessitates the detailed account of two classes of parameters in the model: one regarding the sustainability conditions that have to be respected in order for the coupled system to sustain its functions and a second one quantifying the relative weight society places on consumption. The objective function of society is then intertemporally maximized to yield optimal values for activities, stocks and (shadow) prices. These optimal values of the parameters drive the system through a transitional phase of adjustment ('traverse') to reach a steady state where all (shadow) prices are constant over time. As Winkler notes, the time path of the (shadow) prices is driven by the dynamics of the relative scarcities of land, labour and wilderness ('bison') under the influence of the parametrically fixed 'value system'. For example, if society does not value wilderness explicitly, the shadow price of wilderness reflects its value as the habitat of bison only. Thus, the shadow price of wilderness is the shadow price of farmland minus the cost it takes to turn it into farmland (Winkler 2006a, p. 7-8).

The simple pre industrial model is kept intact in Winkler (2006b), where the author sets out to relax his strong assumptions about society's ability to predict the full range of parameter evolution in the future. What's new is the introduction of unpredictable, novel change in the form of preference evolution, system complexity and ignorance. Novel change takes the form of changes in both the parameters and the functions of the system. A sudden change in preferences, i.e. in the relative welfare weights for consumption, means that the relevant parameters of the model are not constant any more, but change over time. The possibility of deriving accurate estimates of values is no longer guaranteed: '*Frankly speaking, in the presence of genotypic change, the 'correct' valuation of the different stocks over time is in principle impossible*' (Winkler 2006b, p. 98). The only way out seems to lie with practising some sort of adaptive management where plans are revised and updated constantly.

To summarize, different variants of integrated socio-ecological simulation models attempt to capture the intricacies of a dynamic approach to value estimation. These approaches share a role in pushing our understanding of complex systems and alerting both researchers and policy makers to the dangers of oversimplification. A focal point for debate, however, is whether this is an adequate analysis of what is actually happening, or a normative argument for what societies, that is a benevolent planner, ought to be doing. Indeed, the reviewed models are normative in nature in the sense that they describe how the complex socio-ecological systems should evolve over time in order to fulfil the requirements of efficiency and sustainability. This is stressed clearly by Winkler in the two papers mentioned above[23] who states that '*the shadow prices derived from the analysis of a centralized economy, as done in this paper, can be utilized to achieve the optimal set and timing of actions in a decentalized economy…*' (Winkler 2006b, p. 103).

### IV.3.2 Dynamic bioeconomic approaches

Defined simply, a bioeconomic model is one that seeks to maximize some measure of economic value subject to resource dynamics (Conrad 1995). Bioeconomic models are today almost entirely developed within the realm of fisheries economics. As such, they model the optimality and sustainability of management options to regulate fisheries with or without access control, dependency of fishing stocks to water quality, and the process of adjustment by which an optimal stock size is attained (for an overview see Willen 1985, Swallow 1994, Eggert 1998). The latter factor in particular, i.e. the process of adjustment by which an optimal stock size is attained, characterizes bioeconomic models as dynamic in contrast to static bioeconomic models where no such adjustment processes are taken into account (Knowler 2002). In dynamic models of coastal habitat-fishery linkages, for example, changes in wetland area affect the biological growth function of the fishery within a multi-period harvesting model (Barbier 2003).

Dynamic bioeconomic models are of special interest for the present review because they depict the dynamic behaviour of ecosystem value estimates through time. This is especially true for those applications which investigate the indirect benefits derived from regulatory and habitat functions (Barbier 1994). The benefits attributed to these services arise through their support or protection of activities that have directly measurable values. This is tantamount to delineating these services as (environmental) inputs to the production of market goods. As any input in a production process, ecosystem services could be valued in relation to their enhancing the productivity of economic activities. In dynamic approaches, the wetland support function is included in the intertemporal bioeconomic harvesting problem. Any welfare impact of a change in this function can be determined in terms of changes in the long term equilibrium conditions of the fishery or in the harvesting path to this equilibrium (Barbier 2000; 2007; Freeman, 2003).

In a number of publications, E.D. Barbier and his co-authors investigated extensively the dynamic bioeconomic linkages between supporting mangrove coastal wetland services and shrimp fisheries. In Barbier and Strand (1998), an open access fishery model was developed to account explicitly for the effect of deforestation policies in the mangrove area of Campeche (Gulf of Mexico) on carrying capacity and thus production of shrimp fisheries. The basic model consists of two equations: one which modelled the change in shrimp stock over time and a second which modelled the adjustment of fishing effort. The stock adjustment equation includes the mangrove area as a proxy for the supply of supporting services, besides the stock of the previous period. Under the assumption of open access, the fishing effort adjustment

[23] '*Indeed, this is a normative approach in the sense that society has to agree on how it wants to assign weights to different activities, and which restrictions it wants to impose on its future behaviour*' (Winkler 2006a, p. 85).

equation let fishing effort in period t depend on the profits realized in period t-1. The long term open access equilibrium of the shrimp fishery is then estimated.

Using estimates for the economic parameters of shrimp price and cost, the authors are able to simulate the comparative static effects of a change in mangrove area on equilibrium harvest and gross revenue of the shrimp fishery over the 1980–90 period of analysis. On average over the 1980–90 period, a marginal decline in mangrove area (1 $km^2$) produces a loss of 14.39 metric tons of shrimp harvest and US$139,352 in revenues from the Campeche fishery each year. This is equivalent to a reduction of 0.19% in the annual harvest and revenues of the fishery. Since mangrove deforestation occurred at the rate of around 2 $km^2$ annually over the simulation period, the resulting loss each year amounts to about 28.8 metric tons, or US$278,704. The authors conclude that '*given the relatively small rate of annual mangrove deforestation in the Laguna de Terminos over the 1980–90 period, the resulting loss in shrimp harvest and revenues does not appear to have been substantial*' (Barbier and Strand 1998, p. 160). It is interesting to note, that the economic losses associated with mangrove deforestation, or equivalently, the value of the mangrove habitat in supporting the Campeche shrimp fishery, appears to be affected by the level of exploitation.

Barbier et al. (2002) continue in the same spirit to highlight another important result of their model: the fact that the value of an ecosystem service is dependent on the management regime under which it is exploited. This can be seen from two different perspectives. On the one hand, if an open access fishery is more heavily exploited in the long term, the subsequent economic losses associated with the destruction of natural habitat supporting this fishery are likely to be lower (Barbier and Strand 1998). Intuitively, this makes sense: the economic loss caused by a poorly managed business shutting down is less than the economic loss caused by a thriving business shutting down. The share of this economic loss that is attributable to the ecological support function of natural habitat will therefore also be smaller (Barbier and Strand 1998, p. 162; see also Freeman 1991). On the other hand, if the demand for fish has a finite elasticity, then the welfare effects associated with a change in a supporting coastal habitat will vary with the magnitude of the elasticity. For an open access fishery, the changes in consumer surplus associated with habitat-fishery linkages will vary inversely with the elasticity of demand (Freeman 1991). Table 3 illustrates the range of welfare estimates depending on assumed demand elasticity.

A smaller, but important, analytical contribution in the area of dynamic bioeconomic models is provided by the newly designated use of computable general equilibrium models (CGE) as a theoretically correct representation of the economic subsystem. Finnoff and Tschirhart (2008) link a dynamic economic computable general equilibrium (CGE) model with a dynamic general equilibrium ecosystem model (GEEM) to access the welfare consequences of endangered Steller sea lion recovery measures on the Alaskan economy. The potential is demonstrated by estimating the value of marine mammals as an ecosystem service to tourism. The simplified economy is modelled as having three production sectors: the fishery F, recreation and tourism R, and composite goods C. The model was run for a time period of 100 years. The authors use the annual equivalent variations as a measure of welfare changes for any single period across two policy scenarios (30% and 170% fishing quotas in relation to 1997). Cumulative welfare changes and mean annual ecosystem valuations per percentage change in marine mammals for alternative quota rules are displayed in Table 4. On average each one percent annual change in marine mammals in relation to the reference is worth roughly US$ 110,000 within the confines of the model.

**Table 3: Dependence of welfare estimates on elasticity of demand in mangrove-fishery linkages in Thailand.**

| Demand Elasticiy | Marginal value of a change in mangrove area (US$ per ha)[1] | | | Economic value of annual loss of 3,000 ha of mangrove area (US$)[2] | | |
|---|---|---|---|---|---|---|
| | Demersal fish | Shellfish | All fish | Demersal fish | Shellfish | All fish |
| –0.1 | 12.49 | 122.95 | 135.4 | 37,556 | 370,637 | 408,193 |
| 0.5 | 7.65 | 49.23 | 56.88 | 22,972 | 147,967 | 170,939 |
| –1 | 5.15 | 28.14 | 33.29 | 15,465 | 84,505 | 99,970 |
| 1.5 | 3.88 | 19.70 | 23.58 | 11,656 | 59,141 | 70,797 |
| –2 | 3.12 | 15.15 | 18.27 | 9,352 | 45,487 | 54,839 |
| 2.5 | 2.60 | 12.31 | 14.91 | 7,809 | 36,956 | 44,765 |
| –5 | 1.43 | 6.36 | 7.79 | 4,279 | 19,071 | 23,350 |
| 7.5 | 0.98 | 4.28 | 5.26 | 2,947 | 12,851 | 15,798 |
| –10 | 0.75 | 3.23 | 3.98 | 2,247 | 9,691 | 11,938 |

[1] Calculations assume an initial equilibrium quantity demand and price based on observed data for each Gulf of Thailand fishery in 1993. For demersal fish this is harvested output of 5,908,000 kilograms (kg) and price of US$1.51/kg, and for shellfish 15,215,000 kg and US$2.58/kg.
[2] Over 1990–1993, the average annual loss of coastal mangroves in the Gulf of Thailand was estimated to be around 30.22 km2, or approximately 3,000 hectares(ha).
Source: Barbier et al. (2002).

**Table 4: Marine mammal valuation (Equivalent variation, million 1997 US$).**

| Value of leisure | Quota rule | Discounted cumulative welfare change | Mean annual welfare change % in marine mammal inputs |
|---|---|---|---|
| 100 | 30% | 16.52 | 109,626 |
| | 170% | 26.90 | 114,458 |
| 75 | 30% | 16.53 | 109,677 |
| | 170% | 26.91 | 114,493 |
| 50 | 30% | 16.54 | 109,728 |
| | 170% | 26.92 | 114,529 |

Dynamic bioeconomic models help us understand supply-driven dynamics of ecosystem value estimates from a similar perspective to the integrated models analyzed in section IV.3.1. They show the development of relative scarcities of ecosystem services in the socio-ecological system and accordingly the accounting or shadow prices associated with these scarcities. The evolution of welfare changes due to ecosystem service losses is calculated therefore on the basis of such optimum prices.

## V. Conclusions and recommendations

The present review addresses the issue of the dynamics of economic values and preferences for ecosystem goods and services. The issue of ecosystem valuation is addressed within the conceptual framework of the Millennium Ecosystem Assessment. The importance of value magnitudes for policy making is not disputed worldwide. The desirability though of dynamic value estimates for environmental decision-making is more acute in those cases where conflicts between short term and long term options for ecosystem use are present.

Ecosystem valuation is a contested area of research; the issue of dynamics complicates the subject further by injecting aspects of uncertainty, intergenerational justice, discounting, cultural and evolutionary processes of preference evolution into issues of proper value elicitation. Existing evidence on the temporal reliability of WTP estimates seems to indicate a rather stable value and preference structure in the short to medium term. This changes as we move to address long term valuation settings. The recognition of ecosystem complexity and the consequent need for modelling of the dynamics of preferences and values can be partially fulfilled by integrated and dynamic bioeconomic models where the supply-driven dynamics of optimal prices can be calculated. These models though are purely normative and therefore lack any descriptive, positive content.

Demand-driven dynamics focus on human behaviour and societal processes. Such societal uncertainty covers aspects of future preferences, needs and incomes. However, it is plausible to assume that in the field of environmental choices even present preferences and needs are often fuzzy and unarticulated. The act of eliciting present preferences and needs, especially through stated preferences approaches, is much criticised in this respect, as blurring the process of eliciting existing preference structures with that of constructing (or even imposing!) them. The validity of this argument has to be examined on a case by case basis since existing state preference studies do not exhibit a uniform approach to the design and implementation of their questionnaire surveys. Nevertheless, since individual choices are based on incomplete knowledge and uncertainty, they are often bound to be incomplete choices based on partial segments of the full choice set, which can improve with learning. Learning more about these issues is best served by further developing our economic valuation tools for it is exactly through research on the economic valuation of environmental assets that a considerable number of longstanding neoclassical assumptions about preferences and values have been empirically tested and theoretically revised. The fact remains that complexity of ecosystems and societies does not cancel out the need for hard choices in the face of both natural and societal uncertainties (Skourtos et al. 2005).

Do available methods stand up to the task? Several authors make the point that the complexities of dynamic socio-ecological systems need to be better understood and examined within an economic valuation framework. Mixing of methods and pooling of data seems the only way forward. In this respect, the potential of systematic and formalised interdisciplinary research lies in the integration of insights, methods and data drawn from natural and social sciences. This includes recognition of thresholds and other non-linearities in the provision of ecosystem services. In this field, a lot has to be described and much more still has to be learned.

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