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Reviewing the dynamics of economic values and preferences for ecosystem goods and services

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Abstract The present paper addresses the issue of 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 was identified as both demand-driven and supply-driven value dynamics. A survey of temporal reliability tests revealed considerable differences in the time span examined (2 weeks to 20 years). The evidence shows that ecosystem value estimates, as expressed through mean Willingness To Pay remain significantly stable in the time span of 2 weeks to 5 years, but this is not the case for time periods of 20 years. For longer periods, both a weak and strong version of preference evolution were 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 were 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.

Keywords Ecosystem services · Ecosystem values · Environmental preferences · Non-market valuation · Dynamics of values · Integrated models · Test–retest

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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 becomes thus the terrain where a number of controversial analytical decisions in ecology and economics have to be taken and, therefore, an extensive and multifaceted arena of hypothesis testing and empirical validation of basic assumptions and approaches.

The present paper addresses the issue of a dynamic approach to biodiversity conservation by focusing on published evidence on the dynamics of economic values and preferences for ecosystem goods and services. The key questions addressed are "How do human preferences and values for ecosystem services change through time?" and "Do we have the methods and data necessary to assess these changes accurately?" A review of valuation studies of ecosystem services has been undertaken with a specific interest on dynamics aspects of preference formation and value estimates. There has been much work and many reviews on the use of economic, preference-based approaches to estimating the social value of ecosystem services. Recent methodological reviews include: Turner et al. (2003), Balmford and Rodrigues (2008), Fisher et al. (2008). Value estimates for specific ecosystem functions are presented in: 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.

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 Millennium Ecosystem Assessment 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 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 practices.

A digression on dynamics, uncertainty and information gaps

The apparent intervention failures in environmental protection to-date, manifested in the persistence and perseverity of the environmental challenges, are rooted mainly in the complexity and unusual long time-dimensions of ecosystem processes (Vanderwalle et al. 2008). The usefulness of investigating the dynamics of environmental preferences and values is therefore greatly enhanced if attention is drawn to the need for a long-term, adaptive conservation strategy. 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' (Crook 2003, p. 8).

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; Loewenstein and Thaler 1989). 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. The rest of our analysis is concerned with a less 'authentic', though more practical way of analysing dynamic phenomena, the so-called comparative static approach to the dynamics of value estimates.

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 p, income y and ecosystem services q_0:

$$V = u_{max} = v(p, q_0, y) \tag{1}$$

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:

$$E = y_{min} = e(p, q_0, u_0) \tag{2}$$

The compensating variation CV 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, WTP (minimum willingness to accept compensation, WTA), in order to secure a gain in ecosystem service provision (accept a loss in ecosystem service provision). CV can accordingly be expressed as the difference between two expenditure functions achieving the same utility level with different levels of service provision:

$$CV = e(p, q_0, u_0) - e(p, q_1, u_0) \quad (3)$$

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

$$CV = e[p, q_0, v(p, q_0, y)] - e[p, q_1, v(p, q_1, y)] \quad \text{or} \quad CV = g(p, q_0, y) - g(p, q_1, y) \quad (4)$$

We denote the right hand side of Eq. 4 as the variation function s:

$$S = s(p, q_0, q_1, y) \quad (5)$$

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 CV. 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:

$$S_t = s(p_t, q_{t0}, q_{t1}, y_t, D_t) \quad (6)$$

Assuming a linear functional form for $s(\cdot)$, 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:

$$\Delta WTP = s(p_0, q_{00}, q_{01}, y_0, D_0 : \beta_0) - s(p_1, q_{10}, q_{11}, y_1, D_1 : \beta_1) \quad (7)$$

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 and the long-term variability of values. 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 profile D remain constant.

For the purposes of the present review, economic approaches to ecosystem valuation have been categorized into two classes:

- Methods that treat ecosystem services as (final) objects of consumption.
- Methods that treat ecosystem services as (intermediate) 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).

Empirical evidence

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. 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. In cases where a paper addressed multiple values and services, a single citation provided more than one entry in our database. 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.

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 p, y, D and β in the variation function (7). We turn our attention first to 'test–retest' reliability applications before we focus on their long-term evolution.

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" (Arrow et al 1993, p. 4609). As Carson et al. (1997) noted, the reasoning underlying the Panel's call for the "temporal averaging" of WTP responses obtained from contingent valuation 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 1 presents a summary of 19 contingent valuation studies, which address the problem of temporal reliability of value and/or bid function estimates. The studies are presented in ascending order of time span considered: Half of the studies examine a time span of less than a year; 9 out of 20 examine time spans of 1–5 years and only one ventures into a time span of 20 years. Almost half of the studies address (9 out of 20) addresses the temporal stability of recreational values whereas the rest concern health effects and regulating services. 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

Table 1 Temporal reliability studies

Authors	Test	Time elapse between applications	Good/service	Temporal stability
Kealy et al. (1988)	Paired differences of WTP	2 weeks	Chocolate bars	Confirmed: no significant differences
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
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
McConnell et al. (1998)	Equality of WTP. No carryover effect	2 months	Recreational fishing	Confirmed: temporal stability of WTP
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
Brouwer (2006)	Paired *t*-tests for equality of WTP	6 months	Health effects	Confirmed: temporal stability of WTP and bid functions
Reiling et al. (1989, 1990)	Equality of WTP between distinct samples	On season versus off	Black fly control in Maine	Confirmed: WTP stability
Loomis (1989)	Paired *t*-tests for equality of WTP	9 months	Water levels in Mono Lake, California	Confirmed: coefficients and WTP stability
Berrens et al. (2000)	Equality of WTP functions	1 year	Instream flows	Confirmed: temporal stability of WTP and bid functions
Downing and Ozuna (1996)	Equality of WTP functions	1 and 2 years	Recreational fishing	Not confirmed: both coefficients and mean WTP differ substantially
Teisl et al. (1994)	Correlation between responses	12 months	Moose hunting permits	Confirmed: high correlation
Carson et al. (1997)	Equality of parameters between distinct samples	2 years	Oil spill protection	Confirmed: stability of equation parameters
Jakus et al. (2006)	Paired *t*-tests for equality of WTP	4 years	Recreational hunting	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

Table 1 continued

Authors	Test	Time elapse between applications	Good/service	Temporal stability
Whitehead and 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
Brouwer and Bateman (2005)	Equality of average WTP and stability of main WTP determinants	5 years	Flood protection and conservation of wetland habitats	Not conclusive: mean WTP differ significantly but underlying explanatory models remain stable
Zandersen (n.d.)	Paired *t*-tests for equality of WTP	20 years	Forest recreation	Not confirmed: both coefficients and mean WTP differ substantially

Source: Adapted from McConnell et al. (1998)

and find that WTP estimates are significantly different over a 5-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. 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 2-month period. Brouwer and Bateman (2005) employ the same survey instrument and sampling procedure to test for the temporal stability of unadjusted average WTP amounts in a 5-year period. The evidence based on the 20 CV applications in Table 1 is rather conclusive: ecosystem value estimates, as expressed through mean WTP, remain significantly stable in the time span of 2 weeks to 4 years. For a time period of 5 years, the evidence is not conclusive. As it is expected though, value estimates differ substantially for a time period of 20 years.

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 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,[1] 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. In a dynamic setting, ignorance about the qualities of ecosystem goods and services, combined with uncertain supply conditions in the future, prompt individuals either to postpone transformation of natural resources (i.e. delay consumption) or, if agreed to consume now, demand a "commitment cost" (an element of quasi-option value in their WTP). Thus, "WTP includes commitment cost and is a dynamic measure that may change daily as consumers update their information about the surplus the good might yield them."[2] Deep-seated disagreement prevails on the proper

[1] 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).

[2] Corrigan et al. (2003, p. 2). The idea was first analyzed in the context of dynamic valuation inconsistencies by Crocker and Shogren (1993).

exchange rate among competing generations (Groom et al 2005; Viscusi 2007; Dasgupta 2008). The present generation may be future averse or future prone, therefore exhibiting a range of time preference rates. In a well-functioning, optimizing economy the proper discount rate should reflect the opportunity cost of capital (Portney and Weyant 1999). In an imperfect economy though, with future uncertainties and ethical conflicts, the discounting rate has to be derived from an overall conception of intergenerational well-being and the consumption forecast (Dasgupta 2008, p. 150).

A rather neglected but genuine dynamic problem of values and preferences concerns the assessment of *future* preference values for *future* stocks (Horowitz 2002). This *preference evolution* problem is distinct from the temporal stability of values because of the longer time periods involved and the fact that neither the form of utility function nor 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 WTP amounts are positively related to income increases causing an upward shift in the demand schedule of a representative consumer (the so-called Engel curves. See Green 1971). 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 (Braga and Starmer 2005). 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? (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.[3]

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

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

discussed ‘anomalies’ in the neoclassical theory of individual choice, 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 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. 1998, 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.

Supply-driven value dynamic

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.[4]

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’Alessandro 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 develop 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. Unit values of the ecosystem services examined were approximated on the basis of expert judgements and replacement cost, and 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

[4] ‘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).

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. 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 (1983–2010). The simulations point towards a critical dependence of economic value (direct and indirect income derived from the park) on ecological health indices. 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, 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.

In Winkler (2006b) 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 (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.

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. 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. This suggests that if an open access fishery is more heavily exploited in the long term, then subsequent welfare losses associated with the destruction of natural habitat supporting this fishery are likely to be lower. Intuitively, this makes sense. The economic value of an overexploited fishery will be lower than if it were less heavily depleted in the long term. The share of this value 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).

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 poor 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

Table 2 Dependence of welfare estimates on elasticity of demand in mangrove-fishery linkages in Thailand

Demand elasticity	Marginal value of a change in mangrove area (US$ per ha)[a]			Economic value of annual loss of 3,000 ha of mangrove area (US$)[b]		
	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

Source: Barbier et al. (2002)

[a] 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 kg and price of US$1.51/kg, and for shellfish 15,215,000 kg and US$2.58/kg

[b] Over 1990–1993, the average annual loss of coastal mangroves in the Gulf of Thailand was estimated to be around 30.22 km^2, or approximately 3,000 ha

therefore also be smaller. 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 2 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 (CGE) models as a theoretically correct representation of the economic subsystem. Finnoff and Tschirhart (2008) link a dynamic economic 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, recreation/tourism, and a composite consumption good. 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 3. 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.

Dynamic bioeconomic models help us understand supply-driven dynamics of ecosystem value estimates from a similar perspective to the integrated models. 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

Table 3 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

of welfare changes due to ecosystem service losses is calculated therefore on the basis of such optimum prices.

Conclusions and recommendations

The present review addresses the issue of the dynamics of economic values and preferences for ecosystem goods and services. 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, and 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 can guide conservation policy rather as sustainability benchmarks and long-term characterisations of ‘windows of sustainability’.

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 current preferences and needs are often fuzzy and unarticulated. The act of eliciting preferences and needs, especially through stated preference 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 stated 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 (Chavas 2000; Pethig 1994). 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. For the time being though, positive analysis of ecosystem value dynamics could only be based on comparative approaches such as temporal reliability tests.

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References

Arrow K, Solow R, Portney PR, Leamer EE, Radner R, Schuman H (1993) Report of the NOAA panel on contingent valuation. Fed Regist 58(10):4601–4614

Aversi R, Dosi G, Fagiolo G, Meacci M, Olivetti C (1999) Demand dynamics with socially evolving preferences. Ind Corp Change 8:353–408

Balmford A, Rodrigues A (2008) Review on the economics of biodiversity loss: scoping the science. Final report to the European Commission

Barbier EB (1994) Valuing environmental functions: tropical wetlands. Land Econ 70(2):155–173

Barbier EB (2000) Valuing the environment as input: applications to mangrove-fishery linkages. Ecol Econ 35:47–61

Barbier EB (2003) Habitat-fishery linkages and mangrove loss in Thailand. Contemp Econ Policy 21(1):59–77

Barbier EB (2007) Valuing ecosystem services as productive inputs. Econ Policy 22:177–229

Barbier EB, Strand I (1998) Valuing mangrove-fishery linkages: a case study of Campeche, Mexico. Environ Resource Econ 12:151–166

Barbier EB, Strand I, Sathirathai S (2002) Do open access conditions affect the valuation of an externality? Estimating the welfare effects of mangrove-fishery linkages. Environ Resource Econ 21(4):343–367

Bateman IJ, Burgess D, Hutchinson WG, Matthews DI (2008) Learning design contingent valuation (LDCV): NOAA guidelines, preference learning and coherent arbitrariness. J Environ Econ Manag 55:127–141

Berrens RP, Bohara AK, Silva CL, Brookshire D, McKee M (2000) Contingent values for New Mexico instream flows: with tests of scope, group-size reminder and temporal reliability. J Environ Manag 58:73–90

Bisin A, Verdier T (2001) The economics of cultural transmission and the dynamics of preferences. J Econ Theory 97:298–319

Bowles S (1998) Endogenous preferences: the cultural consequence of markets and other economic institutions. J Econ Lit 36:75–111

Braga J, Starmer C (2005) Preference anomalies, preference elicitation and the discovered preference hypothesis. Environ Resource Econ 32(1):55–89

Brouwer R (2006) Do stated preference methods stand the test of time? A test of the stability of contingent values and models for health risks when facing an extreme event. Ecol Econ 60:399–406

Brouwer R, Bateman IJ (2005) Temporal stability and transferability of models of willingness to pay for flood control and wetland conservation. Water Resour Res 41(3):W03017

Carson R, Hanemann WM, Kopp R, Krosnick J, Mitchell R, Presser S, Rudd P, Smith VK (1997) Temporal reliability of estimates from contingent valuation. Land Econ 73:151–163
Chavas J-P (2000) Ecosystem valuation under uncertainty and irreversibility. Ecosystems 3:11–15
Chopra K, Adhikari SK (2004) Environment development linkages: modeling a wetland system for ecological and economic value. Environ Dev Econ 9:19–45
Conrad JM (1995) Bioeconomic models of the fishery. In: Bromley DW (ed) The handbook of environmental economics. Blackwell, Oxford, pp 405–432
Corrigan JR, Kling CL, Zhao J (2003) The dynamic formation of willingness to pay: an empirical specification and test. Center for Agricultural and Rural Development, Iowa State University. Working paper 03-WP 327
Crocker TD, Shogren JF (1993) Dynamic inconsistency in valuing environmental goods. Ecol Econ 7:239–254
Crook S (2003) Sustaining Europe's dynamic coasts. Natl Wetl Newslett 25(1):7–17
Cumberland JH (1991) Intergenerational transfers and ecological sustainability. In: Costanza R (ed) Ecological economics: the science and management of sustainability. Columbia University Press, New York, pp 355–366
D'Alessandro S (2007) Non-linear dynamics of population and natural resources: the emergence of different patterns of development. Ecol Econ 62:473–481
Dasgupta P (2008) Discounting climate change. J Risk Uncertain 37:141–169
Downing M, Ozuna T (1996) Testing the reliability of the benefit function transfer approach. J Environ Econ Manag 30:316–322
EFTEC (2005) The economic, social and ecological value of ecosystem services. Final report to DEFRA. EFTEC, London
Eggert H (1998) Bioeconomic analysis and management. Environ Resource Econ 11:399–411
Epp DJ, Gripp SI (1994) Reliability of contingent valuation estimates for an unfamiliar good: tropical rain forest preservation. Unpublished mimeo
Finnoff D, Tschirhart J (2008) Linking dynamic economic and ecological general equilibrium models. Resour Energy Econ 30:91–114
Fisher B, Turner K, Zylstra M, Brouwer R, de Groot R, Farber S, Ferraro P, Green R, Hadley D, Harlow J, Jefferiss P, Kirkby C, Morling P, Mowatt S, Naidoo R, Paavola J, Strassburg B, Yu D, Balmford A (2008) Ecosystem services and economic theory: integration for policy-relevant research. Ecol Appl 18(8):2050–2067
Freeman AM (1991) Valuing environmental resources under alternative management regimes. Ecol Econ 3:247–256
Freeman AM III (2003) The measurement of environmental and resource values: theory and methods. Resources for the Future, Washington, DC
Green HAJ (1971) Consumer theory. Penguin, Harmondsworth
Groom B, Hepburn C, Kountouri Ph, Pearce D (2005) Declining discount rates: the long and the short of it. Environ Resource Econ 32:445–493
Heal G (1986) The intertemporal problem. In: Bromley DW (ed) Natural resource economics. Kluwer Nijhof, Boston, pp 1–20
Heifetz A, Shannon C, Spiegel Y (2007) The dynamic evolution of preferences. Econ Theory 32:251–286
Higgins SI, Turpie JK, Costanza R, Cowling RM, Le Maine DC, Marais C, Midgley GF (1997) An ecological economic simulation model of mountain fynbos ecosystems. Dynamics, valuation and management. Ecol Econ 22:155–168
Horowitz JK (2002) Preferences in the future. Environ Resour Econ 21:241–259
Jakus PM, Stephens B, Fly JM (2006) Temporal reliability in contingent valuation (with a restrictive research budget). In: Alberini A, Kahn JR (eds) Handbook on contingent valuation. Edward Elgar, Cheltenham, pp 249–262
Jones-Lee M, Hammerton M, Phillips PR (1985) The value of safety: results of a national survey. Econ J 95:49–72
Kaval P (2007) Recreation benefits of U.S. parks. University of Waikato—Department of Economics. Working paper in economics 12/07
Kealy MJ, Dovidio JF, Rockel ML (1988) Accuracy in valuation is a matter of degree. Land Econ 64:158–171
Kealy MJ, Montgomery M, Dovidio JF (1990) Reliability and predictive validity of contingent values: does the nature of the good matter? J Environ Econ Manag 19:244–263
Kettunen M, ten Brink P (2006) Value of biodiversity—documenting EU examples where biodiversity loss has led to the loss of ecosystem services. Final report for the European Commission. Institute for European Environmental Policy (IEEP), Brussels, Belgium, 131 pp

Knowler D (2002) A review of selected bioeconomic models with environmental influences in fisheries. J Bioecon 4:163–181

Ledoux L, Turner RK (2002) Valuing ocean and coastal resources: a review of practical examples and issues for further action. Ocean Coast Manag 45:583–616

Li CZ, Löfgren KG (1998) A dynamic model of biodiversity preservation. Environ Dev Econ 3:157–172

Limburg KE, O'Neill RV, Costanza R, Farber S (2002) Complex systems and valuation. Ecol Econ 41:409–420

List JA (2003) Does market experience eliminate market anomalies? Q J Econ 118:41–72

Liu J, Dietz T, Carpenter SR, Alberti M, Folke C, Moran E, Pell AN, Deadman P, Kratz T, Lubchenco J, Ostrom E, Ouyang Z, Provencher W, Redman CL, Schneider SH, Taylor WW (2007) Complexity of coupled human and natural systems. Science 14:1513–1516

Loehman E, De VH (1982) Application of stochastic modeling to the study of public goods. Rev Econ Stat 64:474–480

Loewenstein G, Thaler RH (1989) Intertemporal choice. J Econ Perspect 3:181–193

Loomis JB (1989) Test–retest reliability of the contingent valuation method: a comparison of general population and visitor responses. Am J Agric Econ 71:76–84

Madureira L, Rambonilaza T, Karpinski I (2007) Review of methods and evidence for economic valuation of agricultural non-commodity outputs and suggestions to facilitate its application to broader decisional contexts. Agric Ecosyst Environ 120:5–20

McConnell KE (1990) Models for referendum data: the structure of discrete choice models for contingent valuation. J Environ Econ Manag 18:19–35

McConnell KE, Strand IE, Valdeãs S (1998) Testing temporal reliability and carryover effect: the role of correlated responses in test–retest reliability studies. Environ Resource Econ 12:357–374

McPeak JG, Lee DR, Barrett CB (2006) Introduction: the dynamics of coupled human and natural systems. Environ Dev Econ 11:9–13

Moran D, Hussain S, Fofana A, Frid C, Paramour O, Robinson L, Winrow-Giffin A (2007) The marine bill—marine nature conservation proposals—valuing the benefits. Final report to DEFRA, London

Norton B, Costanza R, Bishop RC (1998) The evolution of preferences. Why 'sovereign' preferences may not lead to sustainable policies and what to do about it. Ecol Econ 24:193–211

Pethig R (1994) Ecological dynamics and the valuation of environmental change. In: Pethig R (ed) Valuing the environment: methodological and measurement issues. Kluwer, Dordrecht, pp 3–22

Plott CR (1996) Rational individual behaviour in markets and social choice processes: the discovered preference hypothesis. In: Arrow K, Colombatto E, Perleman M, Schmidt C (eds) Rational foundations of economic behaviour. Macmillan, London, pp 225–250

Portney PR, Weyant JP (eds) (1999) Discounting and intergenerational equity. Resources for the Future, Washington, DC

Reiling SD, Boyle KJ, Cheng H, Philips ML (1989) Contingent valuation of a public program to control black flies. North J Agric Resour Econ 18:126–134

Reiling SD, Boyle KJ, Phillips ML, Anderson MW (1990) Temporal reliability of contingent values. Land Econ 66(2):129–134

Scoones I, Leach M, Smith A, Stagl S, Stirling A, Thompson J (2007) Dynamic systems and the challenge of sustainability. STEPS working paper 1. STEPS Centre, Brighton

Shogren JF (2002) A behavioral mindset on environment policy. J Socio Econ 31:355–369

Skourtos M, Kontogianni A, Georgiou S, Turner RK (2005) Valuing coastal systems. In: Turner RK, Salomons W, Vermaat J (eds) Managing European coasts: past, present and future. Springer, New York, pp 119–136

Stern N (2007) The economics of climate change. The Stern review. Cambridge University Press, Cambridge

Stone CD (1974) Should trees have standing? Toward legal rights for natural objects. William Kaufmann, Los Altos

Swallow SK (1994) Renewable and nonrenewable resource theory applied to coastal agriculture, forest, wetland and fishery linkages. Mar Resour Econ 9:291–310

Teisl MF, McCollum DW, Boyle KJ, Reiling SD (1994) Test–retest reliability with independent sample pre-test and post-test control groups. Am J Agric Econ 77:613–619

Turner RK, Paavola J, Cooper P, Farber S, Jessamy V, Georgiou S (2003) Valuing nature: lessons learned and future research directions. Ecol Econ 46:493–510

Vanderwalle M, Sykes MT, Harrison PA, Luck GW, Berry P, Bugter R, Dawson TP, Feld CK, Harrington R, Haslet JR, Hering D, Jones KB, Jongman R, Savorel L, Martins de Silva P, Moora M, Paterson J, Rounsevell MDA, Sandin L, Settele J, Sousa JP, Zobel M (2008) Review paper on concepts of dynamic ecosystems and their services. RUBICODE project. www.rubicode.net

Viscusi WK (2007) Rational discounting for regulatory analysis. Univ Chic Law Rev 74(1):209–246
Walker B, Holling CS, Carpenter SR, Kinzig A (2004) Resilience, adaptability and transformability in social–ecological systems. Ecol Soc 9(2):5. http://www.ecologyandsociety.org/vol9/iss2/art5
Whitehead JC, Aiken R (2007) Temporal reliability of willingness to pay from the National Survey of Fishing, Hunting and Wildlife-Associated Recreation. Appl Econ 39:777–786
Whitehead JC, Hoban TJ (1999) Testing for temporal reliability in contingent valuation with time for changes in factors affecting demand. Land Econ 75(3):453–465
Willen JE (1985) Bioeconomic of renewable resource use. In: Kneese AV, Sweeny JB (eds) Handbook of natural resource and energy economics, vol 1. Elsevier, Amsterdam, pp 61–124
Winkler R (2006a) Valuation of ecosystem goods and services. Part 1: An integrated dynamic approach. Ecol Econ 59:82–93
Winkler R (2006b) Valuation of ecosystem goods and services. Part 2: Implications of unpredictable novel change. Ecol Econ 59:94–105
Zandersen M (n.d.) Valuing forest recreation in Europe: time and spatial considerations. PhD thesis, International Max Planck Research School on Earth System Modelling, Hamburg